Aalto-yliopisto
Insinööritieteiden
korkeakoulu

Kiia Mölsä

Life Cycle Assessment of a Wastewater Treatment and a Sludge Handling Process – Current state and future scenarios

Master's thesis submitted for examination for the degree of Master of Science in Technology.

In Espoo 22.12.2019
Supervisor: Professor Anna Mikola
Advisors: Lic. Sc. (Tech.) Anna Kuokkanen;
M. Sc. (Tech.) Panu Laurell

Aalto-yliopisto
Insinööritieteiden korkeakoulu

Aalto-yliopisto, PL 11000, 00076 AALTO
www.aalto.fi

Diplomityön tiivistelmä

Tekijä Kiia Mölsä

Työn nimi Jätevedenpuhdistus- ja lietteenkäsittelyprosessien elinkaarianalyysi – Nykytila ja tulevaisuuden skenaariot

Maisteriohjelma Vesi- ja ympäristötekniikka **Koodi** ENG29

Työn valvoja Professori Anna Mikola

Työn ohjaaja(t) TkL Anna Kuokkanen, DI Panu Laurell

Päivämäärä 22.12.2019 **Sivumäärä** 79+20 **Kieli** Englanti

Tiivistelmä

Jätevedenpuhdistus suojelee ympäristöä vähentämällä luonnonvesiin päätyvää ravinnekuormaa. Se kuitenkin myös kuormittaa ympäristöä aiheuttamalla päästöjä sekä kuluttamalla merkittäviä määriä luonnonvaroja esimerkiksi prosessissa käytettävän energian ja kemikaalien tuotantoon. Elinkaarianalyysi (engl. life cycle assessment, LCA) on ympäristövaikutusten määrittämiseen tarkoitettu standardoitu menetelmä. Sen avulla voidaan yhdistää prosessin osia tärkeimpiin ympäristövaikutuksiin ja vertailla vaihtoehtoja ja siten hyödyntää sitä päätöksenteon tukena.

Tässä työssä jätevedenpuhdistusprosessille Viikinmäen puhdistamolla sekä lietteen jatkojalostusprosessille Metsäpirtin kompostointikentällä suoritettiin kattava elinkaarianalyysi. Tutkimukseen sisällytettiin myös Viikinmäen prosessin mahdollisia tulevaisuuden muutoksia. Täysimuotoisen elinkaarianalyysin lisäksi prosesseille laskettiin erillinen hiilijalanjälki, jota verrattiin elinkaarianalyysin ilmastonmuutospotentiaalin tuloksiin. Työn tavoitteena oli arvioida, mitkä ovat tärkeimmät Viikinmäen ja Metsäpirtin prosessien aiheuttamat ympäristövaikutukset, mistä tekijöistä ne johtuvat ja miten tulevaisuuden prosessimuutokset vaikuttaisivat tuloksiin.

Elinkaarianalyysissa ympäristövaikutuksia lasketaan useille eri vaikutuskategorioille. Kategorioiden merkittävyyttä toisiinsa nähden vertailtiin normalisoinnin avulla, eli laskemalla kunkin kategorian tuloksen suhteellinen osuus EU:n kokonaistuloksista. Kriittisimmiksi kategorioiksi arvioitiin ne, joiden osuudet EU:n kokonaistuloksista olivat suurimmat. Normalisoitujen tulosten perusteella merkittävimmät ympäristövaikutukset Viikinmäen ja Metsäpirtin prosesseista olivat fossiilisten polttoaineiden loppumisen, rehevöitymisen ja ilmastonmuutoksen edistäminen. Näiden kategorioiden vaikutusta nostivat erityisesti puhdistetun veden typpikuorma, suorat typpioksiduulipäästöt, metanolin tuotanto jätevedenpuhdistusta varten ja turpeen tuotanto lietteen jalostusta varten. Mahdollisista tulevaisuuden muutoksista haitta-aineiden poistoon käytettävän aktiivihiilen tuotanto nosti jätevedenpuhdistuksen ympäristövaikutusta merkittävästi ja saostuskemikaalin vaihto sivutuotepohjaisesta ferrosulfaatista ei-sivutuotteesta valmistettuun ferrosulfaattiin kohtalaisesti. Kuitenkin fossiilisista polttoaineista tuotetun metanolin vaihto biopohjaiseen hiililähteeseen vähensi fossiilisten polttoaineiden käyttöä ja pienensi siten jätevedenpuhdistusprosessin koko ympäristövaikutusta.

Tulosten mukaan Viikinmäen ja Metsäpirtin prosessien aiheuttamia ympäristövaikutuksia voitaisiin merkittävästi pienentää optimoimalla prosessiolosuhteita ja korvaamalla turve ja metanoli muilla vaihtoehdoilla. Herkkyystarkastelun perusteella tulokset olivat luotettavia.

Avainsanat LCA, ympäristövaikutus, jätevedenpuhdistus, lietteenkäsittely, GaBi

Aalto University
School of Engineering

Aalto University, P.O. BOX 11000, 00076 AALTO
www.aalto.fi

Abstract of master's thesis

Author Kiia Mölsä

Title of thesis Life Cycle Assessment of a Wastewater Treatment and a Sludge Handling Process – Current state and future scenarios

Master programme Water and Environmental Engineering **Code** ENG29

Thesis supervisor Professor Anna Mikola

Thesis advisor(s) Lic. Sc. (Tech.) Anna Kuokkanen, M. Sc. (Tech.) Panu Laurell

Date 22.12.2019 **Number of pages** 79+20 **Language** English

Abstract

Wastewater treatment protects the environment by reducing the sewage nutrient load in natural waters. However, it also harms the environment by producing emissions and consuming vast amount of resources in for example the production of energy and process chemicals. Life cycle assessment (LCA) is a standardized method for quantifying these environmental impacts. It can be conducted to relate process components with relevant environmental impacts and compare alternatives in order to steer decision-making.

A comprehensive LCA was conducted for the current processes of Viikinmäki wastewater treatment plant and Metsäpirtti composting facility responsible for the further processing of sludge. The study also covered possible future scenarios of Viikinmäki wastewater treatment process. Additionally, a separate carbon footprint calculation was conducted using an Excel-based tool and compared with the full LCA global warming potential results. The goal of this study was to estimate the most important environmental impacts from Viikinmäki and Metsäpirtti processes, identify the factors causing them and assess how the future scenarios would affect the result.

In an LCA, environmental impacts are calculated for different impact categories. The relevance of these categories was compared using normalization by calculating the share of each category result out of the EU total impact. The categories with the highest shares were considered the most relevant when interpreting the results. According to the normalized results, the most significant potential environmental impacts caused by Viikinmäki and Metsäpirtti processes were eutrophication, global warming, and depletion of fossil fuels. The factors contributing most to these potential impacts were nitrogen load in effluent, direct nitrous oxide emissions, production of methanol for wastewater treatment, and production of peat for composting. In possible future scenarios, environmental impact was increased substantially by activated carbon production for micropollutant removal and also moderately by changing the precipitation chemical from a byproduct-based to non-byproduct-based ferrous sulphate. However, replacing methanol produced from fossil fuels with a bio-based carbon source decreased the consumption of fossil fuels and therefore reduced the total environmental impact.

The results showed that by optimizing operational conditions and replacing peat and methanol with other alternatives, the environmental impact of the wastewater treatment and sludge handling processes could be decreased remarkably. Sensitivity analyses conducted indicated good reliability of the results.

Keywords LCA, environmental impact, wastewater treatment, sludge processing, GaBi

Foreword

This Master's Thesis was conducted at the Viikinmäki wastewater treatment plant, operated by Helsinki Region Environmental Services Authority HSY. The topic of this thesis was initiated by HSY, supporting their vision of a sustainable capital region. Conducting a life cycle assessment is a great way of mitigating the climate change and other environmental challenges ahead of us. HSY is hopefully leading the way also for other companies trying to reduce the environmental impacts of their actions. I would like to express my gratitude for HSY on the opportunity to write this thesis and the funding I have received. I am also grateful for Maa- ja vesitekniikan tuki Ry for supporting my thesis work.

I would like to thank my thesis advisors Lic. Sc. (Tech.) Anna Kuokkanen and M. Sc. (Tech.) Panu Laurell for excellent guidance, encouragement and their constructive comments. Your support throughout this whole process has been crucial. I would also like to thank all my other co-workers at Viikinmäki wastewater treatment plant for their support and for creating such a good and warm atmosphere. I am also thankful for Professor Anna Mikola for commenting and the examining my thesis.

Thank you, Sara for actually reading my thesis. It is not short. Thanks also to my other friends and my family for the much appreciated encouragement and support. The most special thanks belong to Tommi for also reading the thesis and for always bearing with me.

Helsinki 21.12.2019

Kiia Mölsä

Table of Contents

List of Abbreviations and Markings

AAS	Advanced activated sludge process
AD	Anaerobic digestion
AS	Activated sludge
ADP	Abiotic depletion potential
AP	Acidification potential
BOD_7	7-day biochemical oxygen demand
CHP	Combined heat and power
CML	A life cycle impact assessment method
COD	Chemical oxygen demand
DCB	Dichlorobenzene
Eco-Indicator99	A life cycle impact assessment method
Eco-Points97	A life cycle impact assessment method
EDIP	Life cycle impact assessment method
EP	Eutrophication potential
FAETP	Freshwater aquatic ecotoxicity potential
FDP	Fossil depletion potential
FU	Functional unit
GAC	Granular activated carbon
GHG	Greenhouse gas
GWP	Global warming potential
HSY	Helsinki Region Environmental Services Authority
HTP	Human toxicity potential
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LUCAS	A life cycle impact assessment method
MAETP	Marine aquatic ecotoxicity potential
ODP	Ozone depletion potential
PAC	Powdered activated carbon
PE	Population equivalent, in Finland one PE includes a 7-day biochemical oxygen demand of 70 grams
POCP	Photochemical ozone creation potential
R11	Trichlorofluoromethane
ReCiPe	A life cycle impact assessment method
TETP	Terrestric ecotoxicity potential
ThOD	Theoretical oxygen demand
TRACI	A life cycle impact assessment method
WWT	Wastewater treatment
WWTP	Wastewater treatment plant

List of Figures

List of Tables

Introduction

Wastewater treatment (WWT) protects the environment by reducing the sewage nutrient load in natural waters. However, it also harms the environment by producing emissions to air, soil and water and by consuming vast amount of resources in for example the production of energy and process chemicals. Awareness on climate change and for example micropollutants has gained more public attention to the environmental impacts caused by companies and the whole society. This has led to higher interest of companies and public operators on the sustainability of their actions. Also restrictions in regulation are driving organizations to consider the environmental impacts of their processes. As public organizations, water utilities must function according to strict and developing regulations and lead the way to others by making their operations more sustainable. To be able to mitigate the environmental impacts, it is important to properly assess them.

Potential environmental impacts of a process can be quantified with a life cycle assessment (LCA). LCA is an internationally standardized method that combines estimated values with different environmental impact potentials derived from different parts of the process system.

In this study, an LCA was conducted for Viikinmäki wastewater treatment plant (WWTP) treating sewage with 1.1 million population equivalent (PE) from Helsinki capital region, one PE including a 7-day biochemical oxygen demand (BOD_7) of 70 grams. Viikinmäki is currently the largest WWTP in the Nordic countries and it is operated by Helsinki Region Environmental Services Authority (HSY). The LCA was also extended for handling of sewage sludge in HSY's Metsäpirtti composting and soil production facility.

HSY has previously conducted LCAs on their waste management processes and a potable water treatment plant but not on WWT and sludge handling processes. Carbon footprint has been studied previously for the WWT operations, but the calculations have not been comprehensive. For the year 2019, HSY had set an initiative to conduct a full-scale LCA on its Viikinmäki WWT process in order to thoroughly estimate its environmental impacts. This study aims in fulfilling this objective.

The WWT and sludge handling system under study included all the processing steps of wastewater and sludge as well as materials production, energy and transports related to the processes. Infrastructure, equipment, handling of waste and end use of compost and soil products were excluded from the study. In addition to estimating environmental impacts for the current process, possible changes to the process and future treatment steps were studied to compare alternatives and to see what kind of environmental impact further treatment of wastewater would produce. Utilizing the latest available information for a full year of operations, the LCA was conducted with data from year 2018.

Utilizing this study HSY is aiming to both understand the actual environmental impacts its WWT process is causing, and search for ways of reducing them. Accordingly, the research questions of this study were selected as follows:

1) What are the most relevant environmental impacts caused by Viikinmäki and Metsäpirtti processes
2) Which factors in the processes cause these main environmental impacts and how could they be mitigated
3) How would the future scenarios affect the environmental impact of the processes
4) How reliable are the LCA results and
5) How do the results compare to previous research on the topic and could they be generalized

Reviewing the existing literature on LCA, many studies on WWT and sludge treatment processes have been conducted. However, studies on WWT and sludge treatment having both similar process structure and system boundaries, were not found. Some studies, however, have included many similar process parts but with different process configurations. The findings from these studies were utilized in the calculation and comparison of the results of this study.

Many previous studies reviewed also included examination of future changes in the processes. Some of these situations were also investigated in this study while also introducing three new future scenarios for the process under study. These three scenarios were changing the precipitation chemical from by-product based ferrous sulphate to a non-byproduct-based option, changing methanol produced from fossil fuels to a bio-based carbon source and adding a treatment step for reject water deammonification.

This LCA study was conducted utilizing measurement data on gaseous emissions instead of estimating the emissions with emission factors. This differs from the reviewed studies where using measurement data for the gaseous emissions was not indicated. The LCA modelling software used in this study was GaBi the use of which was also not mentioned in the exploited literature. Also, an Excel-based Carbon Footprint Calculation Tool was used to conduct a separate carbon footprint calculation. The carbon footprint result was compared with the results from GaBi.

This thesis will first discuss life cycle assessment in general together with applications in WWT. Earlier research on the topic is then presented and the processes studied are described. After a presentation of the study methods used, the latter part of this thesis forms the LCA following a standardized framework. In the LCA, first the goal and scope of the assessment are defined and the future scenarios are described, followed by the inventory analysis, the life cycle impact assessment and interpretation of the results. Finally, the content of this thesis is summarized with conclusions.

1 Life Cycle Assessment of Wastewater Treatment

In this first chapter, the principles of LCA are presented. Examples of their applications in WWT and possible methods are added for clarity.

1.1 Principles of Life Cycle Assessment

The life cycle of a product or service covers all functions from extraction of raw materials to production, transportation, construction, use, maintenance and eventually recycling and disposal. LCA is a structured method created to evaluate potential environmental impacts throughout the parts or the entire life cycle of a product or a service. Impacts are studied by quantifying the used resources and produced emissions (EC-JRC-IES 2010b, p. iv). LCA offers support and brings environmental perspectives into decision making processes when trying to achieve more environmentally friendly products or identifying environmental indicators and how to measure them. It can also be useful in marketing as environmental aspects are becoming more and more important for consumers (SFS-EN ISO 14040, 2006; Antikainen 2010, p. 12).

An LCA conducted for a product or service can vary depending on the level of detail and system boundaries which are chosen according to the objectives of the study. In an LCA, the term *system* is used for the entire process chain under study, e.g. WWT, and it consists of many processes like sedimentation and sludge dewatering. The most comprehensive form of LCA, known as cradle to grave study (***Figure 1***), covers the life cycle of a product or service from raw material acquisition to recycling and disposal. Some parts of the product's life cycle can be excluded from the analysis with a well justified reason, such as irrelevance of the step for the results. The analysis can then cover the life cycle of the product from raw material extraction to the end-of-life management (cradle to grave), from raw material extraction to the end of the production (cradle to gate), only from the production process chain (gate to gate), or only from a specific part of a product's life cycle (SFS-EN ISO 14040, 2006).

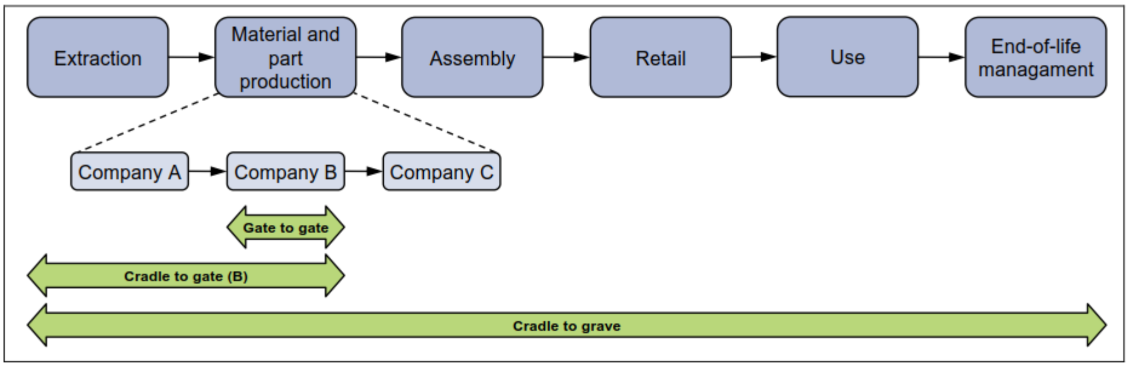

Figure 1: Illustration of a product's life cycle with different extents (EC-JRC-IES 2010b, p.96)

WWT can be considered as an end-of-life process, thus the system under study can also vary from the life cycle approaches presented above. It can be considered as a gate to gate process chain where the system only includes treatment of wastewater until effluent and sludge leave the plant. Then again, it can also be perceived as a process chain from gate to grave if also sludge treatment and reuse as well as effluent load impact to the environment are included within the system boundaries. WWTPs also require infrastructure, which can be either included or excluded from the study depending on the scope.

1.2 Life Cycle Assessment Process

Guidelines for preforming an LCA are given in ISO standards 14040:2006,1044:2006, 14047, 14048:2002 and 14049:2000 which list general guidelines and principles for performing and documenting an LCA. These ISO standards, however, have been criticized for not giving very strict quality specifications for the LCA process and reporting (Antikainen 2010; EC-JRC 2019). The European Commission's Joint Research Centre has compiled two International Reference Life Cycle Data System (ILCD) handbooks to support the LCA procedure giving more detailed instructions for the process with examples (EC-JRC-IES 2010b & 2010c).

SFS-EN ISO -standard (2006a), followed also in this study, lists four main phases of LCA (***Figure 2***). These are:

1) Defining the goal and scope for the assessment
2) Collecting life cycle inventory (LCI) for the process
3) Life cycle impact assessment (LCIA)
4) Interpreting the results

The four phases of LCA are iterative as more specifications might appear during the process. This is indicated in ***Figure 2*** with two-way arrows.

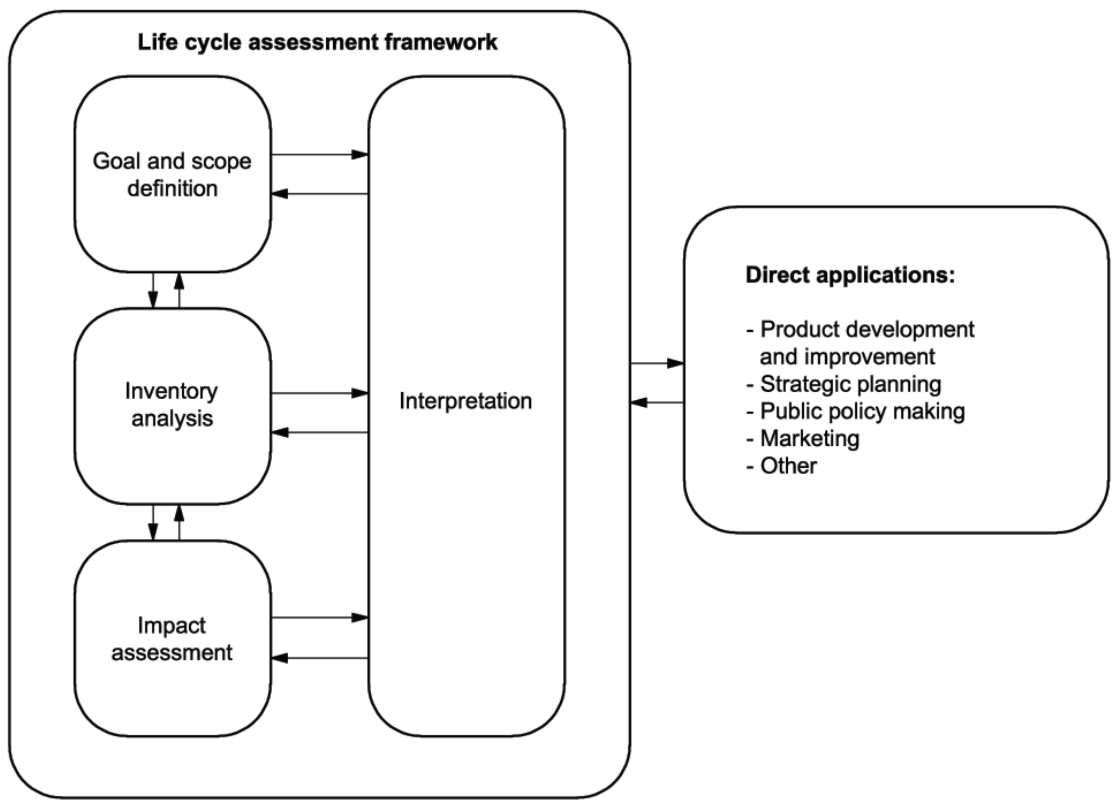

Figure 2: Stages and applications of LCA (SFS-EN ISO 14040, 2006, p. 25)

Defining the goal and scope includes setting the aim, reason and intended use for the LCA while deciding on clear system boundaries for the study. As well as system boundaries, also geographic boundaries should be considered since the site conditions of the product system might influence the results. In this first phase, the assumptions used are listed and a functional unit (FU) and reference flow are chosen. (Guereca *et al*. 2019, p. 5; SFS-EN ISO 14040, 2006, p. 30)

According to Antikainen (2010, p. 17), FU is a reference unit according to which inputs and outputs are normalized. The FU can be for example a certain volume of influent wastewater. FU is used as a reference unit when LCA is used for comparing alternative choices (Guereca *et al*. 2019, p. 5). A reference flow of each process is the amount of product needed to fulfill the FU (SFS-EN ISO 14040, 2006, p. 18). The reference flow is separately defined for each process in the system. For WWT, one reference flow can be for example the amount of precipitation chemical needed to treat the amount of wastewater by the FU.

In the second phase of an LCA, an LCI is compiled for each process within the previously set system boundaries. This means deciding on which processes are to be studied and listing all inputs and outputs for every process. These inputs and outputs are called flows. They include, for example, all materials, energy, water and emissions used or produced in the system. For WWT, examples of inputs are chemicals and energy. Untreated wastewater with nutrients can also be considered as an input for WWT although its nature differs from other inputs used as resources. Examples of outputs from WWT are greenhouse gas (GHG) emissions, sludge and treated water with the remaining nutrients. Flows crossing the system boundaries are known as elementary flows (EC-JRC-IES 2010a, p. 93–94). These are flows coming from nature or exiting the system to nature, for example natural resources and emissions. The European Commission’s Joint Research Centre (2010c, p. 7–8, 14) also points out that compiling LCI data is an iterative process since accuracy in the data might change during the study. Therefore, the inventory data should be clearly listed to be able to change it or to reproduce the study if needed.

An LCIA is used for evaluating the results of an LCI. In an LCIA, the elementary inputs and outputs of an LCI are first classified to contribute to different environmental impacts or damages chosen according to the objective of the study (EC-JRC-IES 2010b, p. 275). In the characterization step, each elementary flow is multiplied by its own specified factor into an impact score. All impact scores in the same impact group are then summed together into one equivalent number (Hauschild & Huijbregts 2015, pp. 7–8), e.g. kilograms of CO_2 equivalent in the group contributing to global warming. Results of an LCIA can also be normalized, grouped and weighted to make their interpretation easier (EC-JRC-IES 2010b, p. 275). With normalization, results from different impact categories are placed into a wider context by comparing them to the estimated summarized results from e.g. EU or the whole world. Setting weighting factors for impact categories again allows summing of all results into single environmental impact scores which can be compared between alternatives.

In the final phase, the results of the LCIA are collected and analyzed according to the goal and scope of the study. To reduce the uncertainty of the study it is important to recognize the most relevant data and results and analyze their accuracy, sensitivity and coherency. Finally, the results sought for in the beginning are presented and recommendations are given for the future (EC-JRC-IES 2010b, pp. 285–286).

On their website the European commission (2019) notes that LCA is a good tool to support a decision-making process but should not be used alone as it only considers environmental aspects leaving out social and economic perspectives.

1.3 Life Cycle Assessment Methods

An LCA can be performed utilizing methods of different extent and goal set at the beginning of the study (SFS-EN ISO 14040, 2006). An LCA can for example only include an LCI phase or one impact category, such as carbon footprint or global warming potential (GWP), or it can contain a study of several impact categories.

Impact categories can be divided to midpoint and endpoint level (***Figure 3***). Investigation of the results on midpoint level offers results as potential impacts divided in different categories such as climate change, toxicity to humans and depletion of materials (Hauschild & Huijbregts 2015, p. 8–9). Some of the most common midpoint impact categories are briefly described in ***Table 1***.

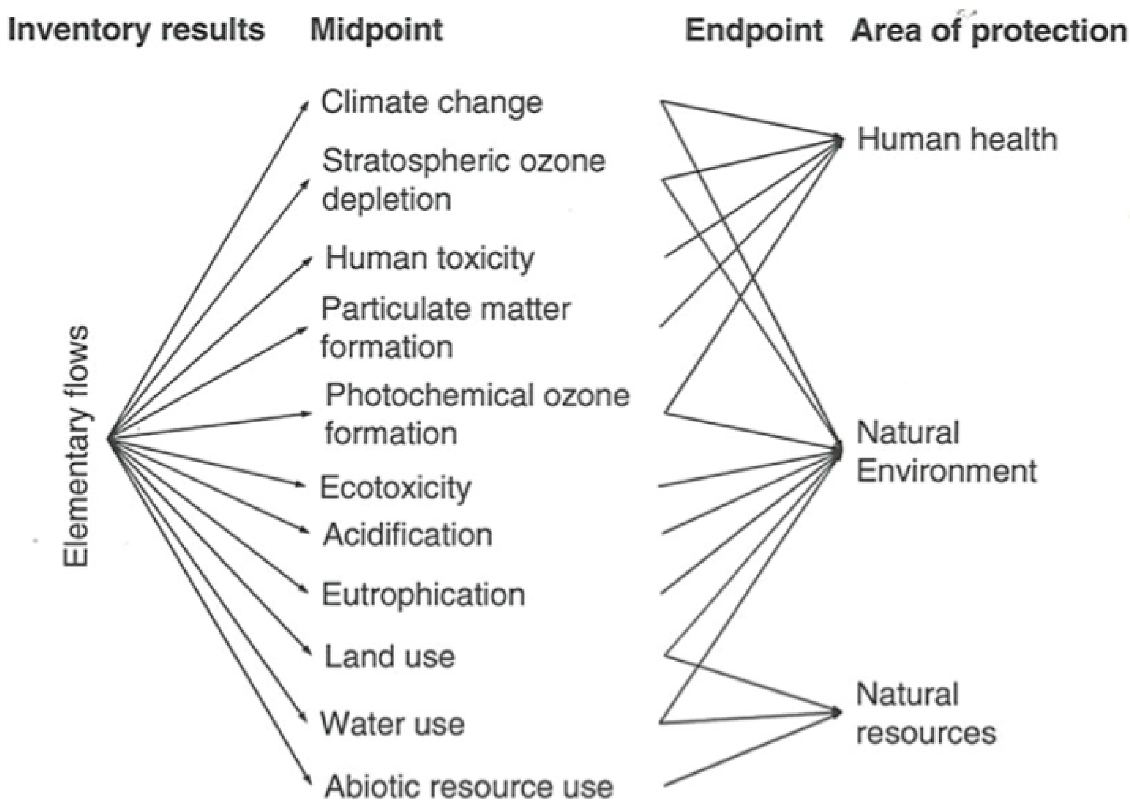

Figure 3: Examples of midpoint environmental impact categories and their contribution to endpoint level areas of protection (Hauschild & Huijbregts 2015, p. 9)

Table 1: Descriptions of common impact categories on midpoint level. Information collected from Tenhunen et al. (2000, p. 38–40), Mattila et al. (2011, p. 8) and van Oers et al. (2002, p. 9–12)

Midpoint impact category	Common abbreviation	Description	Measured equivalent
Climate change/ Global warming potential	GWP	Global warming caused by GHG emissions, comparable to carbon footprint which is often calculated separately without a full LCA	Carbon dioxide
Ozone depletion potential	ODP	Depletion of ozone layer by manufactured chemicals like CFC and HCFC compounds and halons	CFC compound R11
Human- and eco-toxicity potential categories	HTP & ETP	Harm caused by chemicals released to the environment, can be divided to human toxicity and different ecotoxicity categories depending on the type of the receiving ecosystem	Benzene or toluene
Photochemical ozone formation	POCP	Creation of ozone in the lower atmosphere due to reaction of nitrogen with VOC compounds, varies with local climate conditions, disturbs the growth of vegetation and causes health effects when respirated	Ethylene
Acidification potential	AP	Acidification decreases the pH of waters and this way impacts living species	Sulphur dioxide
Eutrophication potential	EP	Increased biomass growth caused by nutrients released to the environment, in natural waters increases oxygen consumption and alters the biodiversity of the ecosystem	Phosphorus or nitrogen
Land use	-	Land use and change in the use of land, causes e.g. GHG emissions and changes carbon cycles and storages, soil quality and productivity, water quality and availability and loss of biodiversity	Area multiplied by time
Abiotic resource depletion potential, often fossil fuels	ADP, FDP (fossil fuels)	Depletion of abiotic resource stocks, often fossil fuels separated	Extracted element, energy content of fossil fuels

These midpoint level categories can be considered as environmental flows between the LCI and endpoint level. For endpoint level analysis, midpoint level impact categories are grouped into three areas of protection for which possible damage is calculated. These areas of protection are human health, ecosystem and natural resources. They are measured in lost healthy years of life, lost species and extra costs to raw materials extraction in the future. (EC-JRC-IES 2010b, pp. 109–110).

According to Antikainen (2010, p. 28–29), besides endpoint and midpoint level study, a third LCA method measures distance to target, for example an emission limit, by calculating eco-factors. It can be applied to substances which are associated with a political guidance or target. The more the current flow of the substance exceeds the political target, the bigger the eco-factor becomes (Frischknecht *et al*. 2009, p. 23).

Different LCIA methods calculate environmental impacts for varying sets of impact categories. Methods use either midpoint or endpoint level or their combination and vary in calculation technique. In these different methods the impacts may also be calculated with different characterization factors and different flows contributing to impact categories (Antikainen 2010, p. 28). The methods are often published especially for a certain context. Some commonly used LCA methods are for example CML, EDIP, Eco-Points97, Eco-indicator99 and ReCiPe for Europe, TRACI for the USA and LUCAS for Canada (Antikainen 2010, pp. 30–31; EC-JRC-IES 2010a, p. 21).

Even though LCIA methods initially include a set of impact categories, some of these categories can be ruled out from the investigation if considered outdated or irrelevant. It is, however, pointed out in the ILCD handbook (EC-JRC-IES 2010b, p. 110) that exclusion of categories should only be carefully considered when interpreting the results as it might lead to restrictions in the conclusions.

As results in different impact categories vary with the method chosen, EC-JRC-IES (2011) has compiled a list of recommended methods for study of different impact categories. The recommendations of methods for LCIA on midpoint level are for example IPCC's 100-year baseline model for GWP, ReCiPe for aquatic eutrophication and CML2002 for resource depletion. No method is recommended in the listing for all categories and some categories even lack a recommended method considered of good quality. LCA being a fairly new field of study, the listing from 2011 can, however, be considered slightly outdated.

As mentioned earlier, an LCA can also be simpler than these comprehensive methods. A more concise study may be easier and faster to perform and still offer directional information of environmental impacts of the process. An LCA study can be based on for example just one impact category or even indicator such as carbon footprint which has been increasingly discussed in media and used by producers. The result can be calculated with measured values and data from literature and companies producing input materials.

1.4 Databases and Software for Life Cycle Assessment

LCA databases can be used to get secondary data for processes and flows that would otherwise be too challenging and time-consuming to compile by the user. There are several both free and commercial databases available which include processes and materials from different fields. The processes added to the model from databases are mainly background processes like electricity or fuel production (Antikainen 2010, pp. 21–22). Datasets offer average or site-specific data that should be checked to fit the production system in question and inventory from a database should only be used if it is more accurate and complete than primary data. (EC-JRC-IES 2010b, pp. 33–35). Some well-known databases are for example the free European reference Life Cycle Database ELCD and commercial Ecoinvent and GaBi. ELCD, however, was discontinued in 2018 (EC 2019).

Performing an LCA by gathering all inputs and outputs and calculating their impacts with factors from a chosen method is possible but it can be very time consuming and complicated. For this reason, LCA modelling software can be used to perform the analysis. A software and a database can either be used separately or a software that already includes one or more databases can be chosen (Antikainen 2010, pp. 22–23). For example, GaBi and SimaPro are commercial software programs. Simapro can be used with several external databases (Simapro 2019) while GaBi includes its own database but can also be used with a few other databases (Thinkstep 2019b). Open LCA is an example of a free modelling software which can be used with several free and commercial databases (Open LCA 2019).

When performing a simpler LCA and for example calculating results for only one impact category, a computation tool like Excel might be more viable to use. An example of this is a Carbon Footprint Calculation Tool developed by Tumlin *et al.* (2014) in their project for Swedish Water which calculates carbon footprints for WWTPs as they produce a significant amount of GHG emissions. This tool is freely downloadable in the internet (VA-teknik Södra 2019).

2 Environmental Impacts from Wastewater Treatment and Sludge Handling

Treating wastewater removes nutrients which would otherwise be released to the environment and this way decreases the environmental impacts caused by wastewater. However, WWTPs consume natural resources and energy while releasing emissions. They also produce big amounts of sludge which needs to be handled. In the next chapters, the resources used, and emissions released from wastewater and sludge treatment are presented. Also, some earlier LCA studies conducted in the field of wastewater and sludge treatment are gathered for comparison and base of this study on Viikinmäki WWTP and Metsäpirtti sludge handling processes.

2.1 Resources to Wastewater Treatment and Sludge Handling

WWTPs, sludge handling facilities and background processes such as energy and chemicals production require raw materials. A WWT process consumes a lot of energy in the form of electricity and heat, the majority of which goes into aeration and pumping (Chae & Kang 2013). Energy is also used in the production of chemicals and other materials. Fuel is used in transportation of chemicals and of end products like sludge and ash. Also composting machinery consumes fuel. Energy and fuel production often require raw fossil resources. Raw materials are used to produce chemicals, compost additives, infrastructure and equipment needed for WWT and sludge handling.

2.2 Emissions from Wastewater Treatment and Sludge Handling

WWT and sludge handling processes release emissions to receiving waters and air. Direct emissions originate from the treatment processes itself while emissions from background processes like chemicals production are considered indirect.

Nutrient compounds causing eutrophication such as organic matter, nitrogen and phosphorous are emitted directly with effluent wastewater as some are present in water during the treatment. Effluent includes also micropollutants. Conventional WWTPs lack treatment steps for specifically micropollutant removal, but some are removed with sludge. New technologies can also remove micropollutants explicitly (Eggen et al., 2014).

The end products of the treatment process, such as sludge or ash, and matter filtered from wastewater also return nutrients to the environment when taken to composting facility and landfills or used in agriculture. From composting facilities and landfills nutrients can escape with stormwaters, which however are often collected and taken back to WWTPs. Agricultural use of sludge decreases the need for extra fertilizer production, but the use of sludge products and fertilizers can lead to release of nutrients to waters. Also, micropollutants adhered in sludge end up in soil (Hospido *et al.* 2010).

A WWT process releases both gaseous and particulate emissions to air. Gaseous emissions from WWTPs are directly formed during wastewater and sludge treatment and on-site energy production. Gaseous emissions originate also indirectly from effluent, external energy production, chemicals production, transports, composting of sludge and landfills. Energy production also releases particulate emissions.

The main gaseous emissions from WWT and sludge handling are GHGs. According to Chen (2019) the three most important GHGs emitted from a WWT process from least to most powerful are carbon dioxide CO_2, methane CH_4 and nitrous oxide N_2O. Methane is a 25 times and nitrous oxide a 298 times stronger GHG than carbon dioxide and both are persistent in the atmosphere. According to Solomon *et al*. (2010), methane stays in the atmosphere around 10 and nitrous oxide 114 years. Often carbon dioxide emissions from WWT and sludge handling processes are excluded from GHG emissions in LCA studies. This is because the Intergovernmental Panel on Climate Change (IPCC) assumes they are all of biogenic origin and thus has left CO_2 out of their GHG guidelines (IPCC 2006, p. 6.6). Tseng *et al*. (2016), however, suggest that part of the CO_2 emissions is of fossil origin as petrochemicals are used in some household products found in wastewater and thus all CO_2 emissions should not be excluded.

Methane and nitrous oxide are also the main GHG emissions released in the sludge composting process (Brown *et al.*2008). In addition, composting releases ammonia gas which is not classified as a GHG but causes harm to for example vegetation (Pagans *et al.* 2005)

Carbon dioxide is inevitably formed in WWT and composting processes when organic material degrades. The amount of other GHG emissions from WWT depends on the operational conditions of the process, such as nutrients concentration and removal efficiency. Nitrous oxide is formed when nitrogen compounds degrade during nitrification and denitrification stages. N_2O emissions vary with dissolved oxygen, nitrite and ammonium concentrations and chemical oxygen demand to nitrogen ratio (Chen 2019). Methane emissions mainly originate from anaerobic conditions when organic matter is removed from water due to methane's low solubility and high mass transfer capacity (Chen 2019). Its formation varies with biodegradable substance, temperature and treatment system type (IPCC 2006, p. 6.7). In a composting process, the conditions where GHG emissions form are much like in a WWT process. Methane is formed in the anaerobic sections of the composting piles (Brown *et al.* 2008) and N_2O in different steps of nitrogen compounds' degradation (Sánchez-García *et al.* 2014). Formation of NH_3 is mainly related to high pH (Pagans *et al*. 2005).

2.3 Earlier Research on Life Cycle Assessment in Wastewater Treatment

2.3.1 Goal and Scope in Earlier Research

The goal and scope in earlier LCA research in wastewater and sludge treatment vary from study of the entire process to examination of a small part of the process. Both gate to gate and gate to grave and their variations have been applied (Corominas *et al.* 2013). Some studies have considered only WWT and sludge treatment processes (Corominas *et al.* 2013), some have included also water collection and reuse (see e.g. Raghuvanshi *et al.* 2017), infrastructure (see e.g. Buonocore *et al.* 2016) or energy production at the WWTP (see e.g. Gustavsson & Tumlin 2013). Also, studying only the sludge treatment process has been popular. This was done by for example Hong *et al*. (2008) and Cao & Pawlowski (2012) who also included infrastructure in their research. Visser *et al*. (2016), in turn, have only studied raw material recovery from wastewater and sludge treatment processes.

Some researchers, for example by Niero *et al*. (2014) and Visser *et al*. (2016) have investigated many different WWTPs. In many studies, alternative processes, configurations or plants were compared (Aaltonen *et al*. 2014; Buonocore *et al*. 2016; Hong *et al*. 2008; Niero *et al*. 2014). Comparison was done to find the environmentally best alternative or to optimize the process in terms of environmental impacts and costs.

The FU chosen was most popularly a specific volume of influent or effluent (Buonocore *et al.* 2016; Corominas *et al.* 2013; Niero *et al.*, 2014; Raghuvanshi *et al.* 2017, Tenhunen *et al.* 2000; Visser *et al.* 2016, p. 5). With studies only looking at sludge treatment, a volume of raw sludge was used as the FU by Cao & Pawlowski (2012) while Hong *et al.* (2008) used a certain mass of dry solids produced. Corominas *et al.* (2013) criticized using volume of wastewater as the FU since it does not consider water quality nor removal efficiency of the plant. The quality of wastewater may vary in great deal with for example different amounts of stormwater led into the system, thus using volume as the FU might lead to unreliable results. There is also possibility of unreliable results with the FU of raw sludge due to changes in sludge amount and quality. Some of the more recent studies in the review by Corominas *et al.* (2013) tackled the issue of changing water quality and volume by using a PE of 5-day biochemical oxygen demand or phosphate as the FU. A PE of BOD_7 was also used by Gustavsson & Tumlin (2013) in their study on Nordic WWTPs' carbon footprints.

Examples of system boundaries and FUs from studied literature are presented in ***Table 2*** for the studies including a WWT process and in ***Table 3*** for studies on only sludge treatment and nutrient recovery from WWT. The tables also include lists of treatment processes studied.

The WWT processes in the LCA studies reviewed included always a biological treatment step, usually with activated sludge (AS) method (see ***Table 2***). Sludge treatment often included anaerobic digestion (AD) and dewatering (Buonocore *et al.* 2016; Gustavsson & Tumlin 2013; Tenhunen *et al.* 2000). Also composting (Aaltonen *et al.* 2014; Hong *et al.* 2008; Raghuvanshi *et al.* 2017; Tenhunen *et al.* 2000), landfilling (Buonocore *et al.* 2016; Tenhunen *et al.* 2000; Hong *et al.* 2008) and agricultural use of sludge (Hong *et al.* 2008; Niero *et al.* 2014; Raghuvanshi *et al.* 2017; Tenhunen *et al.* 2000) were included in some of the studies.

Table 2: Goal and scope in studied literature – Studies including WWT processes

Focus of the study	System boundaries	WWTP and sludge handling processes included	Functional unit	Location	Reference
WWT & sludge handling	WWT and sludge handling processes to grave, including construction, different scenarios, no transports included	AAS*, sludge thickening, belt press dewatering, AD, biogas recovery and on-site energy production, gasification and landfilling of sludge	1000 m^3 of influent	Italy	Buonocore *et al.* 2016
	WWT and sludge handling processes to grave, 16 WWTPs, average calculated, only carbon footprint	Several different treatment methods, all including biological treatment and sludge AD	1000 m^3 of influent	Nordic countries	Gustavsson & Tumlin 2013
	Water and wastewater management system in total including processes and, network infrastructure, 4 WWTPs	AS, AD, dewatering with centrifuges, sludge composting, agricultural use and landfilling	1 m^3 of effluent	Finland	Tenhunen *et al.* 2000
WWT	WWT process at the plant, 4 WWTPs	Advanced biological treatment, chemical precipitation, aerobic digestion and AD, agricultural use and incineration	1 m^3 of influent	Denmark	Niero *et al.* 2014
	WWTP to grave including infrastructure and investments, 4 WWTPs of which some unbuilt	AS, ultrafiltration, chemical precipitation, sludge digestion, energy production on-site, composting	Wastewater treated for 30 years	Finland	Aaltonen *et al.* 2014
	WWT process to gate including collection of wastewater	AS, chlorine, sludge drying, composting and agricultural use	1500 m^3 of treated sewage	India	Raghuvanshi *et al.* 2017

*Advanced activated sludge treatment

Table 3: Goal and scope in studied literature – Studies related to WWT but not including the WWT process

Focus of the study	System boundaries	Treatment processes included	Functional unit	Location	Reference
Sludge treatment	Sludge treatment at WWTP and further handling to grave, including equipment and infrastructure, many different scenarios	Sludge thickening, digestion, dewatering, composting, drying, incineration, melting, landfilling, agricultural use, use of sludge end products as building material	1000 kg of dry solids	Japan	Hong *et al.* 2008
	Sludge-to-energy process from cradle to grave, many different scenarios	AD, pyrolysis	500 m^3 of raw sewage	-	Cao & Pawlowski 2012
Recovery of materials	Recovery process of different materials from WW and sludge treatment	Extraction of phosphate and organic matter with several methods	100 000 PE of influent	The Netherlands	Visser *et al.* 2016

2.3.2 Methods and Data in Earlier Research

In LCA studies conducted on WWT processes some impact assessment methods and databases clearly have been more used than others. In their review, Corominas *et al.* (2013) noted that among 45 WWTPs the most popular methods were CML 2000, EDIP and Ecoindicator99. The recently released ReCiPe was also used more recently for example by Niero *et al.* (2014), Raghuvanshi *et al.* (2017) and Visser *et al.* (2016). Visser *et al.* also calculated single scores for the endpoint results. At least Buonocore *et al.* (2016) and Niero *et al.* (2014) mentioned using suggestions for method from the ILCD handbook. These studies were both conducted on WWTPs in Europe. In the reviewed literature both midpoint and endpoint level studies were conducted. Most researchers using ReCiPe calculated the results for both levels.

Many studies included only some of the impact categories available with the methods chosen according to their relevance. Popular categories included in the reviewed studies were GWP, EP, FDP and AP. Many studies also calculated potentials for photochemical ozone formation, toxicity and ozone depletion. According to a review by Corominas et al (2013), the use of the toxicity impact category in LCA studies was related specially to studies on sludge disposal and micropollutants. Toxicity and eutrophication were often divided in many subcategories. Impact categories used in only one or a few studies were ADP (Visser *et al.* 2016), cancer-causing substances (Tenhunen *et al.* 2000), particulate matter formation (Buonocore *et al.* 2016), heavy metals (Tenhunen *et al.* 2000) and land use (Hong *et al.* 2008). The reasoning for inclusion and exclusion of impact categories in the studies was not usually included. Tenhunen *et al.* (2000, p. 40) mentioned omitting terrestric eutrophication as it was not an issue in Finland. Some studies reviewed by Corominas *et al.* (2013) only conducted the LCI phase.

Calculation of results in different methods varies and hence might give different results. For this reason, the LCA standard (SFS EN-ISO 1044, 2006) suggests performing an uncertainty analysis comparing results from different methods. This uncertainty analysis was only done in a few cases. Niero *et al.* (2014), Pizzol *et al.* (2011) and Renou *et al.* (2008) found that with different methods toxicity potential varied greatly. GWP and resources depletion potential results were similar despite the method used in the studies by Niero *et al.* (2014) and Renou *et al.* (2008). Renou *et al.* also found that the choice of method was not critical for AP and EP. However, Hospido *et al.* (2012) found EP to be the most varying category when using CML, ReCiPe and IMPACT 2002+. Corominas et al. (2013) noted in their review that Eco-points 97 and Eco-indicator 99 seemed to produce results close to each other while being far from results from CML 2000.

The modelling tool was seldom mentioned in the LCA studies reviewed. Simapro was used by at least Visser *et al.* (2016) and Niero *et al.* (2014). Niero *et al.* used the tool for uncertainty analysis with Monte Carlo technique. Gustavsson & Tumlin (2013) used the Excel-based Carbon Footprint Calculation Tool when calculating only carbon footprints to many different Nordic plants.

Clearly the most used database in the literature studied was Ecoinvent (Buonocore *et al.* 2016; Cao & Pawlowski 2012; Niero *et al.* 2014; Raghuvanshi *et al.* 2017; Visser *et al.* 2016, p. 12). In addition, Niero *et al.* used the ELCD database and Hong *et al.* (2008) mentioned using USES-LCA as their primary database in their study on sludge treatment. Some missing data was also taken from literature (Gustavsson & Tumlin 2013; Niero *et al.* 2014).

Only some studies indicated normalizing or weighting their LCIA results (Buonocore *et al.* 2016; Corominas *et al.* 2013; Visser *et al.* 2016, p. 6). Among the reviewed literature, sensitivity analysis was more often performed (Cao & Pawlowski 2012; Gustavsson & Tumlin 2013; Niero *et al.* 2014; Tenhunen *et al.* 2000, p. 66; Visser *et al.* 2016, p. 11; Corominas *et al.* 2013).

2.3.3 Results in Earlier Research

In the following paragraphs, results of LCA studies are presented first by different impact categories and factors causing them. Then, total numerical results from different studies are reviewed.

GWP was measured in the majority of studies reviewed. None of the reviewed studies indicated using measured values for the GHG emissions. Buonocore *et al.* (2016) and Tenhunen *et al.* (2000) mentioned using calculated estimations of the gaseous emissions. In many studies it was found that energy production outside the WWTP for the process needs was the biggest influencer in raising the GWP and fossil depletion potential (FDP) when energy was produced with fossil fuels (Niero *et al.* 2014; Raghuvanshi *et al.* 2017). Producing energy with fossil fuels releases carbon emissions which increase GWP. FDP is impacted as fossil fuels are non-renewable. External energy production was also one of the biggest contributing factors of carbon footprints of Nordic WWTPs calculated by Gustavsson & Tumlin (2013). Another main constituent of the carbon footprints were the N_2O emissions from the WWT processes and GHG emissions from sludge storage before agricultural use. Also, Daelman *et al.* (2013) and Rodriguez-Garcia *et al.* (2012) suggested that N_2O emissions from WWT process were the biggest factor increasing the carbon footprint of a WWTP. According to Niero *et al.* (2014), another important factor in raising GWP and FDP was the production of chemicals, especially $FeCl_3$ used by the plant in the study. The factors causing the rise were not discussed. In addition to these steps, composting contributed to GWP increase as it releases GHG emissions (Aaltonen *et al.* 2014).

Energy production with biogas was calculated to have negative impact on GWP, carbon footprint and FDP (Buonocore *et al.* 2016; Gustavsson & Tumlin 2013; Hong *et al.* 2008; Niero *et al.* 2014; Tenhunen *et al.* 2000). The reason for a negative GWP is considering CO_2 emissions as of biogenic origin and reduced need for energy production with fossil fuels. Aaltonen *et al.* (2014) also pointed out that when sludge is digested, methane is collected for use and this way carbon doesn't end up in the atmosphere as direct emissions. In their study on the Tampere water utility in Finland, Tenhunen *et al.* (2000, p. 56) found that producing heat from biogas at the WWTP generated only a third of the total environmental impact of producing the necessary heat at a heating plant. The reduced need for fossil fuels with energy from biogas naturally also makes FDP negative.

Another factor producing negative GWP and FDP in the studies by Aaltonen *et al*. (2014) Niero *et al*. (2014) and Tenhunen *et al*. (2000) was using nutrients from sludge in agriculture to replace need for extra fertilizer production. Tenhunen *et al*. (2000) calculated that treating and using sludge had more benefits than environmental impacts. However, they reminded that there are also unknown substances in sludge which may harm the environment.

In the studies by Tenhunen *et al*. (2000) and Niero *et al*. (2014), EP was increased by nutrient emissions to sea and fresh waters. According to Niero *et al*. (2014), marine eutrophication was especially increased by released nitrogen. In the model used by Tenhunen *et al*. (2000), nitrogen was also mainly considered as the limiting factor of growth in the sea whereas phosphorous was limiting growth in Finnish inland waters. This means phosphorous emissions would be the biggest factor increasing EP of fresh waters in Finland. Decrease in EP of fresh waters was found by Buonocore *et al*. (2016) when green energy was used instead of fossil fuels. Negative impact was noted in the same study with agricultural use of sludge.

As in many other categories, the use of fossil fuels for energy production was the main factor causing HTP increase (Niero *et al*. 2014). Additionally, iron chemical production, incineration of sludge and heavy metal emissions contributed to the category. As with EP, the use of renewable energy decreased HTP (Buonocore *et al*. 2016). According to Niero *et al*. (2014), a negative impact in the category was again a result of energy production from biogas and agricultural use of sludge. However, some scientists argued that agricultural use of sludge increases HTP (Corominas *et al*. 2013; Hong *et al*. 2008).

Tenhunen *et al*. (2000) found treated and released wastewater to cause AP increase, making the category significant as Finnish ecosystems are sensitive to acidification. Hong *et al*. (2008) also stated that the agricultural use of sludge had an increasing effect on AP.

The above-mentioned impact categories were the most significant and the most widely discussed in the reviewed literature. Energy production and sludge treatment and how they impacted GWP drew the most attention. Some studies compared the environmental impacts of treating wastewater and sludge versus releasing them straight to the environment (Cao & Pawlowski 2012; Corominas *et al*. 2013; Raghuvanshi *et al*. 2017, Tenhunen *et al*. 2000). In most cases, WWT processes with their environmental impacts were found more environmentally friendly than releasing wastewater with nutrients directly to the environment (Raghuvanshi *et al*. 2017; Tenhunen *et al.* 2000; Visser *et al*. 2016). According to calculations by Tenhunen *et al*. (2000), the advantage of a WWT process was tenfold compared to no treatment. In the study by Raghuvanshi *et al*. (2017) also reusing treated wastewater in irrigation was calculated as a negative environmental impact and according to Visser *et al*. (2016) nutrient recovery had more environmental benefits than impacts. The only process treating water more thoroughly which did create positive environmental impact was removal of micropollutants (Høibye *et al*. 2008; Wenzel *et al*. 2008). It was suggested, however, that the characterization factors for micropollutants may be debatable.

The environmental impacts of infrastructure in literature were controversial. According to Lundin *et al*. (2000) and Corominas *et al*. (2013), the significance of infrastructure in the total results varied according to the scale and the systems built. Lundin *et al.* (2000) suggested that for large-scale WWT systems the impact of infrastructure was low compared to operational impacts. Hong *et al*. (2008) found impact from infrastructure low but impact from equipment high.

The results from different impact categories were often interpreted by presenting them with shares of contribution from different process factors (see e.g. Buonocore *et al.* 2016; Gustavsson & Tumlin 2013). Many studies also presented the total numerical results from the entire process. The total results from WWT-scoped LCA studies which announced the total sums were reviewed for perspective. The results from the studies in three popular categories used, GWP, EP and FDP, are presented in ***Table 4***. However, it should be noted that comparison of these numerical results is very limited due to differences in the LCA scopes: The system boundaries, FUs and WWT processes in the studies were different as indicated earlier in ***Table 2***. Also, some of the impact categories were different from each other as for example Raghuvanshi *et al.* (2017) only studied freshwater EP and Niero *et al.* (2014) only marine EP instead of the total result from both.

Table 4: Numerical results for GWP, EP and FDP from reviewed studies including a WWT process. For studies comparing several processes the results presented are calculated averages.

Study by	GWP/Carbon footprint	EP	FDP	FU
Niero *et al.* (2014)	0.17 kg CO_2 eq.	4.8×10^{-3} kg N eq.*	5.0×10^{-2} kg oil	1 m^3 of influent
Buonocore *et al.* (2016)	400 kg CO_2 eq	0.81 kg P eq.	65 kg oil	1000 m^3 influent
Raghuvanshi *et al.* (2017)	1000 kg CO_2 eq.	0.40 kg P eq.**	-	1500 m^3 influent
Gustavsson & Tumlin (2013)	46 kg CO_2 eq.	-	-	PE/year
Aaltonen *et al.* (2014)	1.3×10^8 kg CO_2 eq.	6.6×10^5 kg PO_4 eq.	5.8×10^5 MWh	30 years

* only marine eutrophication
**only freshwater eutrophication

Comparison of the results becomes easier when the results are translated to similar units (***Table 5***). Scaled to 1000 m^3 of influent, the GWP results in studies by Buonocore *et al.* (2016), Niero *et al.* (2014) and Raghuvanshi *et al.* (2017) varied from around 170 to 670 kg CO_2 eq. The EP result was announced in the weight of either phosphorus, nitrogen of phosphate equivalent. Using emission factors from CML method, all EP results were translated into an equivalent of phosphate. The factors used were then 0.42 kg PO_4/ kg N for nitrogen and 3.06 kg PO_4/ kg P for phosphorus. With this translation, the EP results from studies by Buonocore *et al.* (2016), Niero *et al.* (2014) and Raghuvanshi *et al.* (2017) become 2.4, 2.0 and 0.8 kg PO_4 eq./ 1000 m^3 of influent, respectfully. Thus, the EP results also varied but were all of similar scales. Also, the FDP result was of similar scale in studies by Buonocore *et al.* (2016) and Niero *et al.* (2014). The amount of oil was translated into energy content in MJ using a factor of 45 MJ/ kg oil from World Nuclear Association (2019). The differences in the results become bigger when scaled for larger amounts.

Table 5: Numerical results for GWP, EP and FDP from reviewed studies including a WWT process – Translated to similar units when possible

Study by	GWP/Carbon footprint	EP	FDP	FU
Niero *et al.* (2014)	170 kg CO_2 eq.	2.0 kg PO_4 eq.*	2.3×10^3 MJ	1000 m^3 influent
Buonocore *et al.* (2016)	400 kg CO_2 eq	2.4 kg PO_4 eq.	2.9×10^3 MJ	1000 m^3 influent
Raghuvanshi *et al.* (2017)	667 kg CO_2 eq.	0.82 kg PO_4 eq.**	-	1000 m^3 influent
Gustavsson & Tumlin (2013)	46 kg CO_2 eq.	-	-	PE/year
Aaltonen *et al.* (2014)	1.3×10^8 kg CO_2 eq.	6.6×10^5 kg PO_4 eq.	7.0×10^7 MJ	30 years

* only marine eutrophication
**only freshwater eutrophication

2.3.4 Challenges found in Earlier Research

LCA being a very site specific and hypothetical study, the challenges listed in the reviewed literature were often about data quality and the uncertainty of the results (Corominas *et al*. 2013; Niero *et al*. 2014). Corominas *et al*. highlighted that the same data cannot be used everywhere as the factors, pollutants and dynamics vary with location. Therefore, more regional data should be developed. There is also need for methods for studying newly emerging substances like micropollutants. Niero *et al*. (2014) also found challenges especially in modelling end-of-life treatment of sludge. ISO standards were found too loose for coherent LCA studies and they should be developed further (Corominas *et al*. 2013).

Interpreting LCA results and finding the most significant issues may be challenging with a big amount of data especially on midpoint level. According to Corominas et al. (2013), an endpoint level analysis with only three result categories was found easier to understand but was less specific and less reliable due to its more speculative nature. The researchers also found that human health and surface waters related categories were often highlighted above resource categories. Difficulties in results interpretation were also discussed by Niero *et al*. (2014) and Tenhunen *et al*. (2000). According to Tenhunen *et al*., comparing ecological impacts with benefits of the treatment process was problematic while Niero *et al*. struggled to give recommendations as results varied with impact categories chosen.

Both Corominas *et al*. (2013) and Niero *et al*. (2014) stated that the idea of WWT being about removing pollutants should shift to recovery of materials. Utilizing results of LCA properly, according to Corominas *et al*. (2013), requires stronger integration of LCA in the decision-making process and more communication with the decision-makers.

3 Wastewater Treatment and Sludge Handling Process Description

In this chapter, the processes included in the LCA study are described with the main resources and emissions.

3.1 Viikinmäki Wastewater Treatment Process

3.1.1 Viikinmäki Process Introduction

Viikinmäki WWTP was built in 1994 and it treats sewage from the Finnish capital region. The process is based on a conventional activated sludge method and it includes mechanical, biological and chemical treatment of wastewater (***Figure 4***).

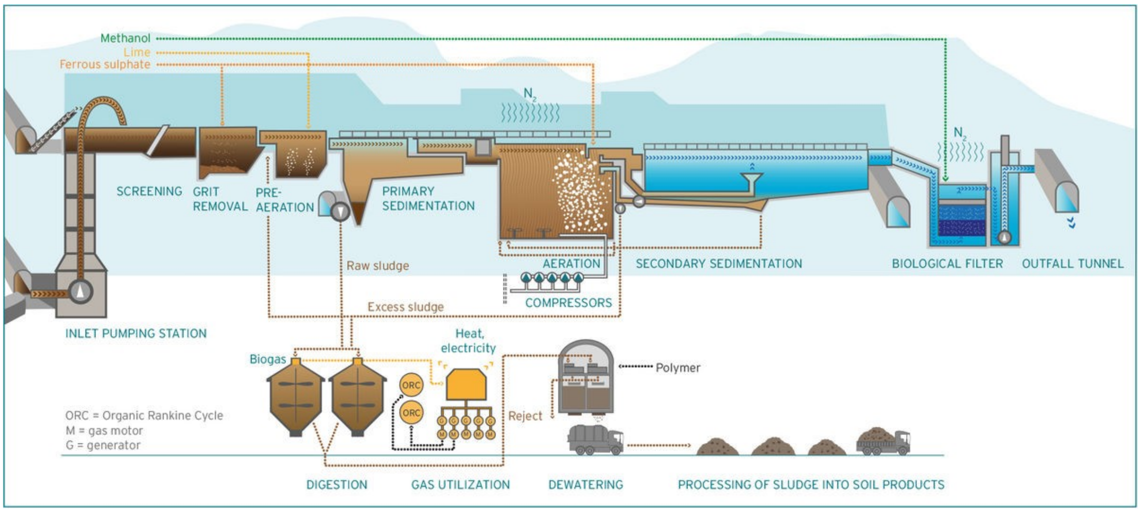

Figure 4: Viikinmäki WWT process

Influent from sewage network is first pumped to mechanical screening of bigger objects which are taken to waste-to-energy plant. Wastewater then continues from screening to removal of sand and floating scum. Removed sand is taken to washing and reuse. The next process step in WWT is pre-aeration where calcium hydroxide is added for control of pH in nitrogen removal. Primary settler then removes roughly 50% of solids, organic matter and phosphorous. Ferrous sulphate is added to the primary sedimentation for precipitation of phosphorous.

After primary sedimentation, the activated sludge process removes nitrogen and more organic matter and phosphorous. Most of the nitrogen is turned into nitrogen gas while some of it ends up in sludge with organic matter and phosphorous. Denitrification occurs first in the anoxic zones of aeration tanks followed by nitrification in aerobic conditions. Sludge is then settled in secondary sedimentation tanks.

Wastewater continues from the sedimentation tank to a biological denitrification filter which enhances nitrogen removal. Methanol is added in this step to provide carbon for denitrifying bacteria. Some organic matter and phosphorous are also removed. Denitrifying (DN) filtration effluent is discharged to the sea via tunnel eight kilometers off the coast of Helsinki.

In peak flow situations some wastewater can be bypassed straight from primary sedimentation to the effluent tunnel. In this case polyaluminium chloride and polymer can be added to the bypassed water to enhance precipitation of phosphorous.

Raw and excess sludge are removed in primary sedimentation and used for energy production. In sludge treatment, sludge is digested in four digestors after which it continues to intermediate storages. Polymer is added to the sludge for dewatering in centrifuges. Dewatered sludge is transported by trucks to the Metsäpirtti facility where it is composted. Part of the compost is used in agriculture. Sand, peat and biotite are added to the remaining compost and turned into soil for domestic use.

Reject water from the centrifuges is recirculated to pre-aeration. Some of the reject water first goes through a biological deammonification treatment.

Biogas from the digesters is collected for energy production. Electricity and heat are generated from biogas in gas engines. Electricity production is enhanced with Organic Rankine Cycle equipment turning excess heat into electricity. Some gas can also be burned in boilers for heat or torched. In addition to electricity from biogas, some electricity is produced with solar panels at the plant.

3.1.2 Load to Viikinmäki Wastewater Treatment Plant

Viikinmäki WWTP receives wastewater via sewage network from 855 000 residents which makes 85% of influent to the plant. The rest 15% comes from industry in the capital region. In 2018 the flow to the plant was on average 250 000 m^3/day. In addition, sewage suction trucks carry liquid waste and sludge from e.g. restaurants and septic tanks to the plant. These liquid wastes are treated in the WWT and sludge treatment processes. The amount and concentration of wastewater to the plant varies with water consumption and weather, as stormwater enters the network diluting the wastewater. If the flow in the sewage network exceeds the maximum capacity of the network system, the surplus water overflows locally. If the flow at the plant exceeds the maximum biological treatment capacity, excess wastewater is directed to bypass the biological process as described in chapter 3.1.1. In 2018, the total flow in the sewage network was 92 000 000 m^3 out of which 57 000 m^3 overflowed before reaching the plant. No bypasses were made at the plant.

The biggest components removed from wastewater are organic matter, phosphorous, nitrogen and solids. In 2018, Viikinmäki WWTP received load for 1,3 million PE. This load does not include sewage suction truck transports from outside the sewage network. Load to the plant is presented more precisely with reduction shares in chapter 3.1.4. Wastewater also includes micropollutants some of which are removed in the process by degradation and with sludge.

3.1.3 Resources Used in Viikinmäki Wastewater Treatment Process

The main resources used in a WWT process are electricity, heat and process chemicals. In addition, fuel is needed for transports and additional materials are needed for soil production.

In the Viikinmäki WWT process, electricity is consumed mainly in aeration, pumping and sludge dewatering. Electricity is also needed for air-conditioning and lighting as the treatment plant is placed underground. Electricity consumption in the process is tied to the influent flow and load.

Viikinmäki WWTP produces most of the required energy with engines and boilers from biogas on site. Some energy is also produced at the plant with solar panels. In 2018, Viikinmäki WWTP had to buy 3% of the electricity from the market. The energy bought was also produced with renewable sources. All electricity used at Viikinmäki WWTP can therefore be counted as green energy. In 2018, all heat consumed at Viikinmäki was produced at the plant. Extra heat from Viikinmäki was even used at Vanhakaupunki potable water treatment plant. Most of the heat was produced with biogas and a small 0.5 % share with light fuel oil.

The chemicals used in the Viikinmäki WWT process are ferrous sulphate $FeSO_4$, calcium hydroxide $Ca(OH)_2$, methanol CH_3OH and polymer. The production of these chemicals requires raw materials and energy. The energy used in chemicals production is not included in the WWTP's consumption above. Ferrous sulphate is, however, a side product from the production of titanium oxide.

Transporting chemicals, sludge and waste for incineration consumes fuel. The consumption of fuel varies with distance and weight of the truck and the load. Polymer is brought to Viikinmäki WWTP also by sea freight and methanol by train. Fuel is also consumed by equipment of the composting facility.

The Viikinmäki WWT process also needs land-, human and monetary resources as well as equipment and construction resources. These aspects, however, fall out of the scope of this study.

3.1.4 Direct Emissions from Viikinmäki Wastewater Treatment Process

After treatment, the effluent from Viikinmäki WWTP contains still organic matter, nitrogen, phosphorous and solids. These are released to the sea with the effluent water. Loads, reduction and emissions of these substances are gathered in ***Table 6***.

Table 6: Viikinmäki WWT process load, reduction and emissions to sea in 2018

Substance	Load to plant (ton/a)	Reduction (%)	Released with effluent (ton/a)
BOD_7	27 152	98	424
COD*	58 677	93	4007
Nitrogen	4875	91	454
Phosphorous	596	97	15
Solids	29 381	99	314

*Chemical oxygen demand

In addition to these substances, wastewater effluent contains small amounts of organic micropollutants and heavy metals.

Direct emissions to air come from the treatment process itself and energy production. The most important gaseous emissions from the treatment process are nitrous oxide N_2O from nitrogen removal and methane CH_4 especially from anaerobic treatment steps. Methane is also released in the beginning of the process as it is formed in the network and occasionally from digestors when the pressure gets too high. Carbon dioxide CO_2 is also released but is excluded in LCA due to its mostly biogenic origin.

Emissions to air from energy production at the plant consist of methane CH_4, carbon monoxide CO, nitrogen oxides NOx, Sulphur oxides SOx, particulate matter and carbon dioxide CO_2. Energy production releases both fossil and biogenic carbon dioxide due to consumption of light fuel oil in support of heat production. Some screening waste is also incinerated at Vantaa Energy waste-to-energy plant contributing to air emissions while also producing energy to the market. End use of waste, however, is not included in this study.

Truck transports and the composting field machinery consume fossil fuels containing carbon and therefore mainly emit carbon dioxide.

3.2 Metsäpirtti Composting and Soil Production Process

3.2.1 Metsäpirtti Process Introduction

The sewage sludge from WWT is handled at the Metsäpirtti facility. The digested and dewatered sludge is composted in windrows of different sizes after mixing with peat and horse manure for better quality. Ratios of mixed substances vary according to the pursued quality of the compost. Also, coffee waste from industry is composted with the mixture.

The windrows are turned with heavy machinery for consistent quality and better composting conditions. Each windrow is composted for at least six months. Biotite and sand are then mixed to a share of the mature compost to form soil products. The sand is used as a mineral needed for good quality soil and the biotite is added for potassium. The soil products and the remaining compost are sold to consumers and agriculture.

3.2.2 Resources Used in Metsäpirtti Process

Metsäpirtti treats sewage sludge from Viikinmäki WWTP and HSY's other WWTP located in Suomenoja, Espoo. Roughly 73 000 tons of sludge is handled in the process yearly, most of which comes from the Viikinmäki plant. Other resources used are peat, horse manure, sand and biotite which are additives for compost and soil.

Metsäpirtti consumes energy in the form of electricity as well as water for a small-scale real-estate and fuel for machinery. As for Viikinmäki, purchased electricity is produced with renewable sources. Machinery at the facility consume light fuel oil.

The transports to and from Metsäpirtti consume diesel. Transports into Metsäpirtti consist of sludge and additives transports. Transports from Metsäpirtti include products transports and waste.

As with the Viikinmäki process, Metsäpirtti also consumes monetary-, human- and land resources which are left outside of this study.

3.2.3 Direct Emissions from Metsäpirtti Process

The composting process releases methane, nitrous oxide and ammonia. Also, carbon dioxide is emitted, mostly of biogenic origin. Transports and machinery emit fossil carbon dioxide.

Wastewater and stormwater gathered from Metsäpirtti process are treated at Viikinmäki WWTP and are thus only included in the Viikinmäki LCA.

4 Implementation of Life Cycle Assessment for Viikinmäki and Metsäpirtti Processes

In this chapter, the LCA process as well as the tools and methods chosen for the implementation are described in brief.

4.1 Life Cycle Assessment Process

An LCA of wastewater and soil production processes was conducted according to the procedure from ISO standards 14040:2006,1044:2006, 14047, 14048:2002 and 14049:2000. The goal and scope were first set including definition of the system boundaries, division of the processes inside the system and determination of the resource flows between processes. These were specified iteratively throughout the LCA process.

The comprehensive life cycle impact analysis was conducted by modelling the current WWT and soil production processes with background processes with GaBi software. GaBi is presented in chapter 4.2. Gabi was chosen in this study as it was already in use in HSY's waste management department. The chosen scenarios were added to the model and the results were calculated with CML2001 method, described in chapter 4.3. The results were calculated in most CML2001 – Jan. 2016 baseline impact categories (***Table 7***). The only excluded category from CML2001 baseline was GWP including biogenic carbon. Only GWP excluding biogenic carbon was used due to IPCC (2006) recommendations on GHGs.

Table 7: Used impact categories from CML2001–Jan. 2016 baseline

Impact category included	Abbreviation	Unit
Abiotic depletion potential of elements	ADP (elements)	kg Sb eq.
Abiotic depletion potential of fossil fuels	ADP (fossil)	MJ
Acidification potential	AP	kg SO_2 eq.
Eutrophication potential	EP	kg PO_4 eq.
Freshwater aquatic ecotoxicity potential	FAETP	kg DCB* eq.
Global warming potential (100 years), excluding biogenic carbon)	GWP	kg CO_2 eq.
Human toxicity potential	HTP	kg DCB eq.
Marine aquatic ecotoxicity potential	MAETP	kg DCB eq.
Ozone layer depletion potential	ODP	kg R11** eq.
Photochemical ozone creation potential	POCP	kg C_2H_4 eq.
Terrestric ecotoxicity potential	TETP	kg DCB eq.

*Dichlorobenzene
** Trichlorofluoromethane

The results from GaBi were imported into Excel where they were analyzed. To compare the results from different categories and to place them into European context, they were normalized according to method CML2001- April 2015, EU25+3, year 2000 excluding biogenic carbon. This means calculating the shares of the different impact category results of the total impacts of EU. The used method already included data of the EU-wide results for 28 EU countries from year 2000. This data was assumed accurate enough as it was the most recent normalization data by CML. A sensitivity analysis was conducted in GaBi using the biggest influencing factors with both local parameter analysis and Monte Carlo analysis. Additionally, the magnitude of the most relevant impact category was calculated with another method, TRACI, and compared with the result from CML. In the local parameter analysis, a 5% negative and positive variation was studied for the biggest influencing factors. Monte Carlo analysis was implemented using 50 simulation runs. According to results, process options were examined.

A carbon footprint was also calculated for the Viikinmäki and Metsäpirtti processes with an Excel-based Carbon Footprint Calculation Tool alongside of the comprehensive LCA. The Carbon Footprint Calculation Tool is described is chapter 4.4. Sensitivity of the results from the Carbon Footprint Calculation Tool was examined by altering minimum and maximum factors included in the tool. The carbon footprint results were then compared with GWP results from GaBi.

4.2 GaBi

GaBi is a commercial LCA modelling software by the German company Thinkstep. It includes its own databases and processes which can be used when primary data is not available. Some of the processes and flows in GaBi are created with German data while others are also available specifically for different countries. Also, Ecoinvent and U.S. LCI databases can be used with the software (Thinkstep 2019b).

In GaBi, processes inside the chosen system boundaries are modelled into a plan. Processes are connected to each other with flows of energy and materials (***Figure 5***). After modelling the entire system with LCI into GaBi, the software calculates results for LCIA. Different LCIA methods can be chosen by the user. GaBi also creates its own automatic report of the LCA. According to Thinkstep (2019a) GaBi has over 10 000 users.

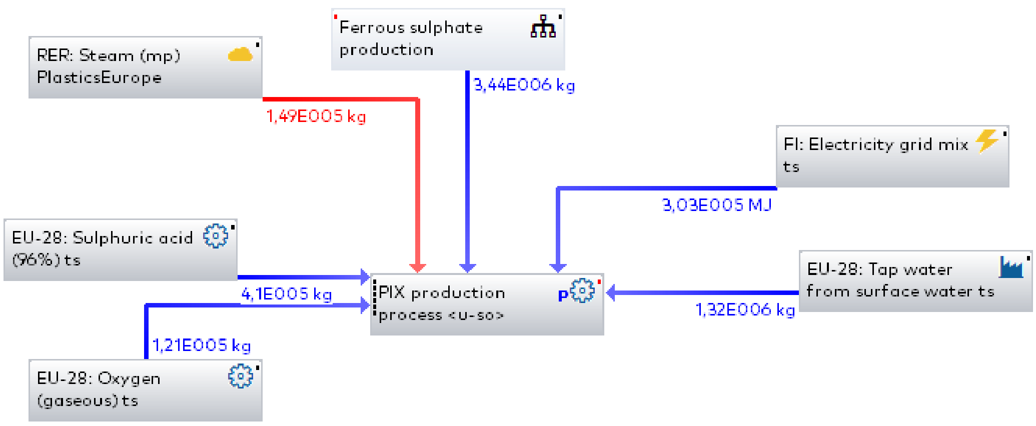

Figure 5: An example of processes and flows connected in GaBi. Processes are presented with grey boxes and flows with arrows.

4.3 CML2001

CML2001 is a common LCA method created in the Netherlands by the Institute of Environmental Sciences in Leiden University (Acero *et al*. 2016, p. 9). CML2001 has been published together with a handbook on LCA (Guinée *et al*. 2002). The method includes midpoint impact categories that can be divided to baseline and non-baseline categories from which baseline is more commonly used (Acero *et al*. 2016, p. 9). Non-baseline group includes a wider list of different impact categories than baseline. CML also offers its own normalization factors for EU and the whole world.

4.4 Carbon Footprint Calculation Tool

An Excel-based Carbon Footprint Calculation Tool was used in comparison with modelling results from GaBi. This tool was developed for Swedish Water Development during a project on carbon footprints of Scandinavian WWTPs by Gustavsson & Tumlin (2013). The model has then been translated to English and is freely available on the website of VA-teknik Södra (2019).

The tool is filled with inputs from a WWTP and it calculates carbon footprints for all functions contributing to emissions. Emission factors are taken from literature and chemical producers and their sources are listed. These footprints are visible in the results and are also grouped to wider functions like WWT, energy production and transports. Results are calculated in CO_2 equivalents. The carbon footprint of the entire plant is also related to load and reduction of BOD, COD, nitrogen and phosphorous.

Some parts of the tool were modified to better describe the Viikinmäki WWT process and especially the Metsäpirtti soil production process. These modifications are described in chapter 6. Both calculations and emission factors were modified when more accurate background data from HSY was available.

The tool was used as a starting point for LCA modelling as it was created for calculating the environmental impact of a WWTP and thus offered a good frame for the study. The results from the Carbon Footprint Calculation Tool were compared with GWP results from GaBi.

5 Goal and Scope

In this chapter, the goal and scope of the LCA are described according to ISO 14040 series standards. Also, the future scenarios studied are presented.

5.1 Goal and Scope of the Current Process Study

The goal of this LCA was to gain knowledge on the environmental impacts of the Viikinmäki WWT process and the Metsäpirtti composting and soil production process in year 2018 and analyze how possible future scenarios would influence the impacts. Year 2018 was chosen for the LCA to get as recent results as possible for a full year. Compared to previous years, nitrogen removal was on the same level, but reduction of phosphorus was higher. The aim was to evaluate which parts and factors in the processes cause the most relevant impacts and if these factors could be changed to cause a lower impact. The changes in LCA results can be evaluated by updating the calculations yearly. The future scenarios were only studied for the complete LCA, excluding them from the separate carbon footprint calculation. This study is published according to Aalto University's master's thesis guidelines. The results are mainly intended for internal use of HSY's WWT but can also be related to other similar processes with caution.

The system in the scope for both LCA and carbon footprint calculation was a gate to gate process of the Viikinmäki WWT and sludge handling processes described in chapter 3. The study included all following processes, presented also in ***Figure 6***:

- Wastewater and sludge treatment process in Viikinmäki WWTP
- Sewage overflows to the environment from network
- Metsäpirtti composting and soil production process
- Production of chemicals needed in Viikinmäki WWT process
- Production of substances added in composting and soil production
- Production and consumption of energy in all processes
- Production and consumption of tap water in all processes
- All transports including
    - Chemical and soil additives transports
    - Waste transports
    - Compost and soil products transports
    - Fuel needed for transports

Infrastructure and equipment were left outside the scope. Lundin *et al.* (2000) suggested in their earlier research that equipment and infrastructure have little effect on LCA results of large-scale WWTPs. Also, collecting LCI for them requires a substantial amount of time. Viikinmäki WWTP was built 25 years ago, hence LCA of the infrastructure would have offered little recommendations contrary to the process itself, which can be modified to some extent. The only scenario with significant changes to the current infrastructure would have been the removal of micropollutants as it would require mining of rock. Besides infrastructure, also the use of compost and soil products and end-of-life treatment of waste from the plants was excluded from the system due to time restrictions and uncertainty of data on end use.

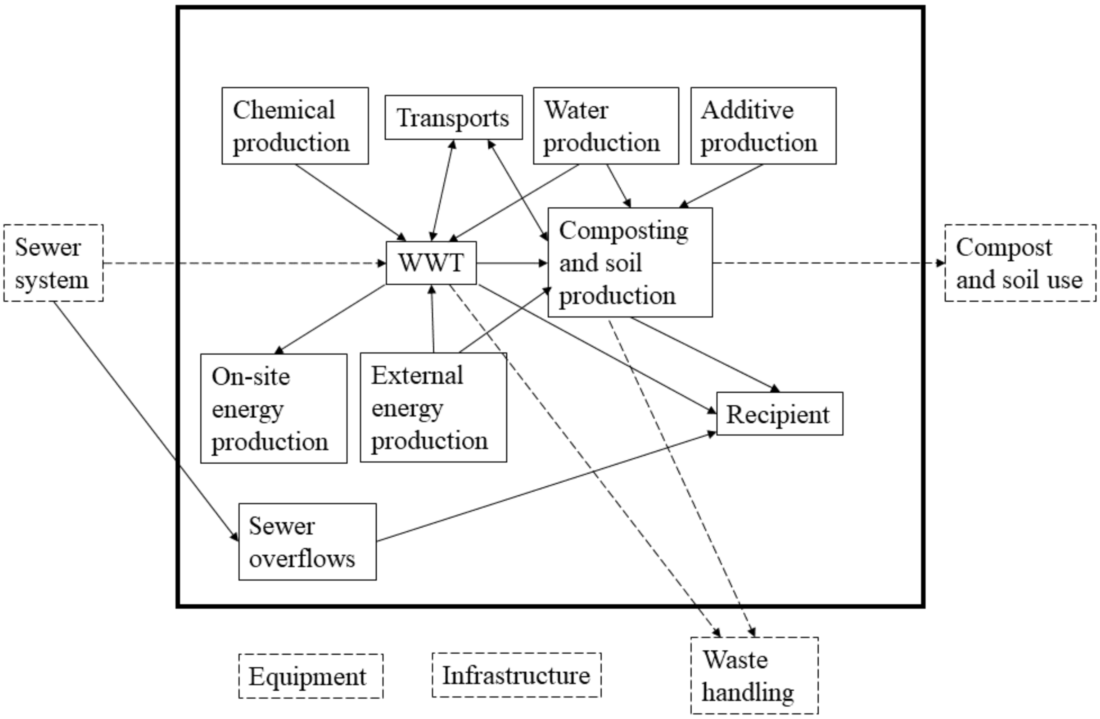

Figure 6: System boundaries

The FU of the LCA was the wastewater inflow to Viikinmäki WWTP in year 2018. With this FU, reported values for flows could be taken straight as inputs for the model and the results would be directly sums for year 2018. Reference flows would then also be the yearly amounts measured. The total results were also scaled to the unit of PE and 1000 m^3 of influent for comparison to future years and other plants.

The method used for LCIA was CML2001 baseline on midpoint level as it was commonly used in previous research. Carbon footprint was also calculated with Carbon Footprint Calculation Tool which did not use any specific LCA method.

The data collected consisted of primary information on elementary flows crossing the system boundary as well as information on process types. Knowing the process types, e.g. types of transport or chemical production processes, leads to secondary information on elementary flows from other sources.

The results of this LCA should be handled as directional amounts rather than precise values for different impacts. This is due to choice of method as well as possible inaccuracies in data due to many estimations. However, uncertainty of the results was assessed in sensitivity and uncertainty analyses.

5.2 Future Scenarios

Besides performing an LCA for the current WWT process of Viikinmäki plant, possible future scenarios were investigated to see how they would impact LCA results. These scenarios are presented in ***Table 8***. The scenarios were based on existing preliminary design made by or for HSY WWT.

In the first scenario, the precipitation chemical ferrous sulphate was changed to different alternatives. Changing the precipitation chemical was studied as a preparation for possible future availability issues of the current chemical. As the current chemical is produced from a byproduct, the first option was to investigate the impact of preparing ferrous sulphate via synthesis route if byproduct was no longer available. Also, the use of another common precipitation chemical, ferric sulphate, was studied. In this scenario, ferric sulphate was produced using ferrous sulphate from byproduct. A third commonly used precipitation chemical, polyaluminium chloride, was not studied due to unavailability of data. Both the production process of precipitation chemical and the amount dosed were changed in scenario 1. The amount of ferrous sulphate stayed the same but amount of ferric sulphate needed was related to difference in iron mass between the different chemicals. The amounts are presented in with the LCI in chapter 6.1.3

In scenario 2 an effluent polishing step for phosphorus removal, including precipitation and disc filtration was added between DN filtration and effluent pumping. The scenario was added due to possible future restrictions in treatment requirements. In this scenario the additional chemicals used were ferric sulphate and polymer. Besides the production of these chemicals, transports and electricity consumption were added to the current process. The estimated decrease in effluent phosphorus was considered in the calculations

The third scenario investigated was adding removal of micropollutants after the current WWT process. As scenario 2, also this scenario was included due to anticipated restrictions in the environmental permit of Viikinmäki WWTP. The micropollutant removal included ozonation, powdered activated carbon (PAC) and treatment of PAC sludge. The process studied was taken from a preliminary design by HSY. The ozone for the process was assumed to be made from oxygen which was produced on site. The collection of PAC sludge was also assumed to consume a similar amount of resources as precipitation and disc filtration in scenario 2. Therefore, scenario 2 was copied as a part of scenario 3. Scenario 3 included the production of chemicals, transports and decreased phosphorous load in effluent. As data of most micropollutants in effluent were often under the limits of quantification, their removal was not included in calculation but was considered when handling the results.

In the fourth scenario, bio-based alternatives for methanol, currently produced from natural gas, were studied for decreasing the environmental impacts caused by the additional carbon source used in DN filtration. As the denitrifying bacteria can only utilize short-chained sources of carbon, ethanol and acetate could be possible substituents for methanol. Due to unavailability of the production process data for bio-based acetate and bio-based methanol, only bio-based ethanol was studied. The consumption of ethanol was estimated from the amount of methanol used, using theoretical oxygen demands (thOD) of methanol and ethanol. The amounts are presented in chapter 6.1.3. This estimation method was used assuming the differences in the thODs of the chemicals were comparable to the differences in the nitrate consumption.

In scenario 5, the deammonification of reject water from dewatering was expanded to all reject from the current 15% share. It was assumed an option with a low environmental impact for lowering the nitrogen load of the WWT process and the operational costs of the plant. Treating all reject water was not assumed to affect the final effluent load but to cause less load to the aeration process. However, deammonification of reject water might have effect on nitrous oxide emissions. According to Li *et al*. (2019) the change in N_2O emissions has varied in different studies on deammonification. Due to this discrepancy, scenario 5 was studied with three different amounts of N_2O emissions: It was assumed that in the current WWT and deammonification processes the ratios of nitrogen removed and N_2O emissions produced were the same. Thus, in the current process roughly 15% of N_2O emissions arose from deammonification. In the three sub-scenarios, the emissions from deammonification were kept the same, doubled and cut in half. As deammonification both consumes electricity and decreases electricity demand in other parts of the WWT process, the total difference in electricity between current state and scenario 5 consumption was assumed negligible.

Table 8: Descriptions of modelled scenarios

Scenario		Sub-scenario	Changes to current process
1	Changing precipitation chemical from ferrous sulphate made from byproduct to	a) Ferrous sulphate made via synthesis route b) Ferric sulphate made from byproduct ferrous sulphate	Changing chemical production process
2	Effluent polishing by adding precipitation of phosphorous and disc filtration		New process step after DN filtration including chemical production, transports, energy use and decreased phosphorus load in effluent
3	Adding micropollutant removal		New process steps after DN filtration: ozonation, PAC, sludge treatment, precipitation and disc filtration; including chemicals, transports and decreased phosphorus load in effluent
4	Changing carbon source of DN filtration from methanol to bio-based ethanol		Changing chemical production process
5	Deammonification of all reject water using different sub-scenarios for N_2O emissions from deammonification	a) no change in emissions b) Double emissions c) Halve emissions	Expanding deammonification to all reject water, decreasing consumption of calcium hydroxide and methanol and changing direct N_2O emissions

6 Life Cycle Inventory Analysis

In this chapter, the principles used when compiling an LCI for the study are presented first for the full LCA and then for the separate carbon footprint calculation. According to EC-JRC-IES (2010c, p. 7–8, 14), an LCI should be clearly listed to be able to change the data and reproduce an LCA study. For this reason, detailed inventories are presented in tables in appendices, referred to in this chapter.

6.1 Inventory for GaBi Model

6.1.1 Principles Used in Collection of Life Cycle Inventory

To ensure the best available quality of data, information was taken from primary sources as often as possible as secondary data is less accurate. Data used was also compared with other sources when possible. Data for the LCI was collected primarily from HSY. Additionally, data from chemical and soil additive producers and transport companies was collected. Secondarily data was taken from GaBi databases and literature, respectively. Data not found from these sources was estimated.

Data from other countries and years was assumed to represent the process in question adequately when country-specific values were not available. Also, data from previous years was expected satisfactory for processes without data from 2018. Data was taken from sources which were thought to best match current Viikinmäki and Metsäpirtti processes out of the available options. Technology and substance mixtures from GaBi databases were thought to produce satisfactory results when true proportions were unidentified.

6.1.2 Wastewater Treatment, Composting and Soil Production

The following mass and volumetric flows were based on HSY's reporting from year 2018 and needed no further estimation:

- Amount of wastewater and sludge treated in the processes
- Amount of chemicals and soil additives used
- Amount of tap water used in the processes
- Amount of compost and soil products sold
- Amount of waste into and from Viikinmäki and Metsäpirtti
- Load from the plant with treated wastewater to the sea
- Emissions to air from the WWT process

In addition to measured gaseous emissions from the WWT process, biogas was emitted from sludge digesters. During foaming incidents there are emissions of biogas form the digesters that are not measured, but an estimate has been made based on pressure, average gas production and duration. GHG emissions from the Metsäpirtti composting process were last measured in 2011. Emissions for year 2018 were quantified for the full LCA based on these measurements. A detailed inventory of elementary flows including changes in different scenarios is presented in ***Appendix 1*** for the Viikinmäki process and in ***Appendix 2*** for the Metsäpirtti process.

The flows of water and sludge circulating between different process steps was estimated from HSY's operating software data. This was done for modelling purposes but did not affect the results as Gabi does not provide information based on non-elementary flows. Non-elementary flows originate from and end up inside the system boundaries and thus they do not contribute to the total environmental impact from the system.

6.1.3 Chemicals

The consumption data for each chemical used in the Viikinmäki WWT process in year 2018 was based on HSY's annual report. To simplify modelling, each chemical was allocated to only one process inside the system contrary to the real situation where the addition of chemicals is distributed to many treatment steps. Again, this did not affect the results of the LCIA as these were not elementary flows. Principles for modelling the production process of each chemical is presented in the paragraphs below. Full life cycle inventories for chemicals without a production process from GaBi databases are listed in ***Appendix 3***.

The production process of ferrous sulphate was not directly available in GaBi as it was produced from a byproduct. Thus, there was also no data for ferric sulphate made from byproduct ferrous sulphate in scenario 1b. Production processes of ferrous sulphate and ferric sulphate were created in GaBi based on information received from the producer (Kettunen 2019). In the process, ferrous sulphate is excavated from a pile using an excavator and a mining truck. These processes were taken from GaBi database and only needed an estimation of excavated mass as inputs. Besides this excavation process, production of ferric sulphate also includes the use of oxygen, chemicals, water and energy. This inventory might lack some emissions or minor inputs, but they were assumed to have little impact on the overall result. GaBi includes production process data for ferrous sulphate produced with synthesis. This process data was used in scenario 1a.

The amount of ferric sulphate dosed in scenario 1b was changed from the current consumption of precipitation chemical due to difference in iron mass between ferrous sulphate and ferric sulphate. According to the datasheets received from the producer, iron mass of ferrous sulphate is 17,5% and iron mass of ferric sulphate is 11%. Consumption of ferric sulphate was estimated using the same mass of iron as dosed with ferrous sulphate.

Production data for calcium hydroxide was taken from GaBi databases as the process model was comparable to the one used for manufacturing calcium hydroxide that was used in Viikinmäki process (Aurola 2019).

LCI for polyacrylamide or other flocculant polymer was unavailable as primary or secondary data. In their research Bonton *et al*. (2011) assumed that LCI of acrylonitrile, the main compound in polyacrylamide production, was close to the LCI of polyacrylamide. Following this assumption, production information of acrylonitrile found in GaBi was used instead of polymer data.

Methanol for Viikinmäki process is produced conventionally from natural gas (Lindholm 2019a). As no primary data from the manufacturing process nor secondary data from GaBi was available, LCI for methanol was taken from a study by Althaus *et al*. (2007). Data for bioethanol was retrieved from GaBi database. The consumption of bio-ethanol was estimated using thODs as it was assumed that the differences in the thODs of the chemicals were comparable to the differences in the nitrate consumption. According to

Pitter & Chudoba (1990, p. 94), the thOD is 1,5 g/g for methanol and 2,09 g/g for ethanol. Amount of ethanol used was estimated using the same mass of thOD as dosed with methanol. This approximation, however, might slightly underestimate the amount of ethanol needed to replace methanol in the process when comparing the ratio with a study by Christensson *et al.* (1994).

No data was found on PAC production. The same was found by Zhang *et al.* (2018) who suggested that the inventory of granular activated carbon (GAC) is close to that of PAC. The main difference in the inventories is the regeneration process not performed for PAC. Hence, the inventory of GAC without regeneration could be used also for PAC. Inventory for GAC was mainly taken from the study by Zhang *et al.* except for transportation processes which were changed to better describe the scenario. The consumption of PAC was estimated in existing preliminary design made by or for HSY WWT.

Oxygen production for ozonation process was taken from GaBi databases. Data for ozone production from oxygen was received from HSY's drinking water treatment plant. As for PAC, the consumption of ozone was estimated in existing preliminary design plans.

6.1.4 Soil Additives

The consumption data of soil additives used in Metsäpirtti composting and soil production process in year 2018 were based on HSY's annual reporting. Principles for modelling the production processes of additives used are presented in the paragraphs below. Full life cycle inventories for additives without a production process from GaBi databases are listed in ***Appendix 4***.

The production process for sand was taken from GaBi databases. Biotite for Metsäpirtti process was produced from a by-product. No suitable secondary inventory data for biotite was found but the production process was built in the GaBi model with information from Juntunen (2019). According to Juntunen, biotite is dewatered with centrifuges and burned before transportation. The process consumes light fuel oil and electricity. The amount of light fuel oil was known, and the oil manufacturing process was taken from GaBi databases. The consumption of electricity was estimated based on electricity demand of centrifuges at Viikinmäki WWTP and weight of the product. Emissions from biotite burning were modelled with an incineration process from GaBi database. This inventory contains many estimations, but they were assumed comprehensive enough considering the overall result.

LCI for the peat production process was taken from a study by Boldrin *et al*. (2010). The original LCI included production, transportation and use phases but only the production inventory was used as more accurate primary data for transports and use in soil production were available. Horse manure lacked further handling before transportation to Metsäpirtti process and did not therefore need production inventory data. A process for sand production was found from GaBi databases.

6.1.5 Transports

Truck transportation distances and payloads for chemicals, sand waste and screenings, sludge, soil additives and compost and soil products were taken from HSY's reporting. Transportation distances for rail transports, polymer transportation and PAC transportation by ferry were estimated with a map. Distances for other waste and external liquid waste transports were estimated to be roughly 30 km each way since they mainly come from inside the capital region. Transport processes for the alternative chemicals in different scenarios were modelled same as the original chemical due to lack of data. PAC sludge in scenario 3 was assumed to be transported to Vantaa Energy waste incineration plant. Thus, a copy of screenings transport process was used for PAC sludge.

In the GaBi model, payloads for each truck transport process were cut in half to model a return trip with the vehicle going back empty. Ferry and rail transports were modelled only one way as they were assumed to have payload both ways. It was assumed that all road transports utilized diesel as their fuel. According to Lindholm (2019b) the train carrying methanol from Russia was mainly electric but also diesel-engined. EU-wide diesel production process for the transports was taken from GaBi databases. Electricity for rail transports was modelled as an EU-average process since a specific process was unavailable for Russian electricity grid mix. It was assumed that polymer and PAC were transported by ferry using light fuel oil with an EU-average production process.

Only distances and payloads of GaBi road transport processes were altered. For rail transports, also shares of electricity and diesel consumption were changed to better represent the real situation. When the European emission standard for a transportation process was unknown, a default mix from GaBi was used. Also shares of rural and urban road and type of diesel were left on default settings as they were assumed to cause little effect on results. LCI for transports with external sources listed is presented in ***Appendix 6***.

6.1.6 Energy

Electricity and heat consumed, and emissions produced in on-site energy production in the current Viikinmäki and Metsäpirtti processes were listed on HSY's annual reporting and could be directly modelled in GaBi. As electricity purchased by HSY is certified eco-friendly, the production process of electricity was modelled using renewable energy resources found in GaBi databases. As the true production methods were unknown, three sources of renewable energy used in Finland were chosen. These were wind, biogas and biomass. They were assumed to have an equal share of the total purchased electricity. The process data was taken from GaBi databases. Amounts of energy consumed is included in the appendices ***Appendix 1*** and ***Appendix 2*** for Viikinmäki and Metsäpirtti respectfully with the process inventories.

Inputs and outputs of the on-site energy production process in Viikinmäki are presented in ***Appendix*** *7*. The inventory also includes scenario 5, where three different possibilities of nitrous oxide emissions were modelled as there is no certainty of how reject water treatment would alter the emissions: The N_2O emissions from reject water treatment were cut in half, kept the same and doubled. The only elementary input to the energy production process at the plant was light fuel oil used in boilers.

Electricity consumption of ozonation was estimated with data from HSY's Vanhakaupunki drinking water treatment plant. Precipitation and disc filtration was assumed to have roughly the same electricity demand with DN filtration and electricity for pumping wastewater into micropollutant removal process was assumed the same as the current effluent pumping. As in dewatering in biotite production, electricity demand for dewatering of PAC sludge was estimated from the current sludge dewatering process in ratio to the amount of sludge treated. Changes in the total electricity consumption in different scenarios is included in the inventory of the Viikinmäki process in ***Appendix 1***.

6.2 Inventory for the Carbon Footprint Calculation

Inputs for the carbon footprint calculation were mainly the same as for the comprehensive LCA. The Carbon Footprint Calculation Tool initially included emission factors from literature which were used for almost all processes. Modifications made to the emission factors in the tool and differences in inventories between the methods are presented in the paragraphs below.

In the carbon footprint calculation, the GHG emissions produced in the composting process were calculated in the model by default and no primary data was used. Contrary to the system boundaries used in the GaBi model, also the effects from end use of the sludge products were considered in the Carbon Footprint Calculation Tool by default.

Soil additives were not originally considered in the Carbon Footprint Calculation Tool but were added separately using emission factors already listed in the tool for peat and sand. The emission factors for biotite and horse manure were missing from the tool. They were assumed zero for coherency since also another side product used, ferrous sulphate, was assumed to have zero impact in the tool.

In the carbon footprint calculation, emission factors for different transport trucks in the tool were changed to more precise ones from HSY. Also, a ferry transport emission factor from HSY and a train transport emission factor from literature were added. These factors suited better the vehicles used for HSY transportations than the ones initially in the tool. The factors used, however, were from year 2009. Return trips of empty trucks were added as the tool originally only included one-way trips. The emission factors used for transports are listed in ***Appendix 5***.

7 Life Cycle Impact Assessment With GaBi

In this chapter, an overview of the process models built in GaBi is first presented. The results of the LCIA are then first presented in total and later described in more detail. The results are mainly expressed using percentages of contribution from process factors to different impact categories. This is because the total results from different impact categories cannot be compared with each other due to different units.

7.1 The Process Models

Models of the processes at Viikinmäki and Metsäpirtti were built according to life cycle inventories. Overview of the Viikinmäki process model is presented in ***Figure 7*** and Metsäpirtti process model in ***Figure 8***. Also, production of eco-electricity, methanol, ferric sulphate and PAC were modelled in their own plans and connected to Viikinmäki process model.

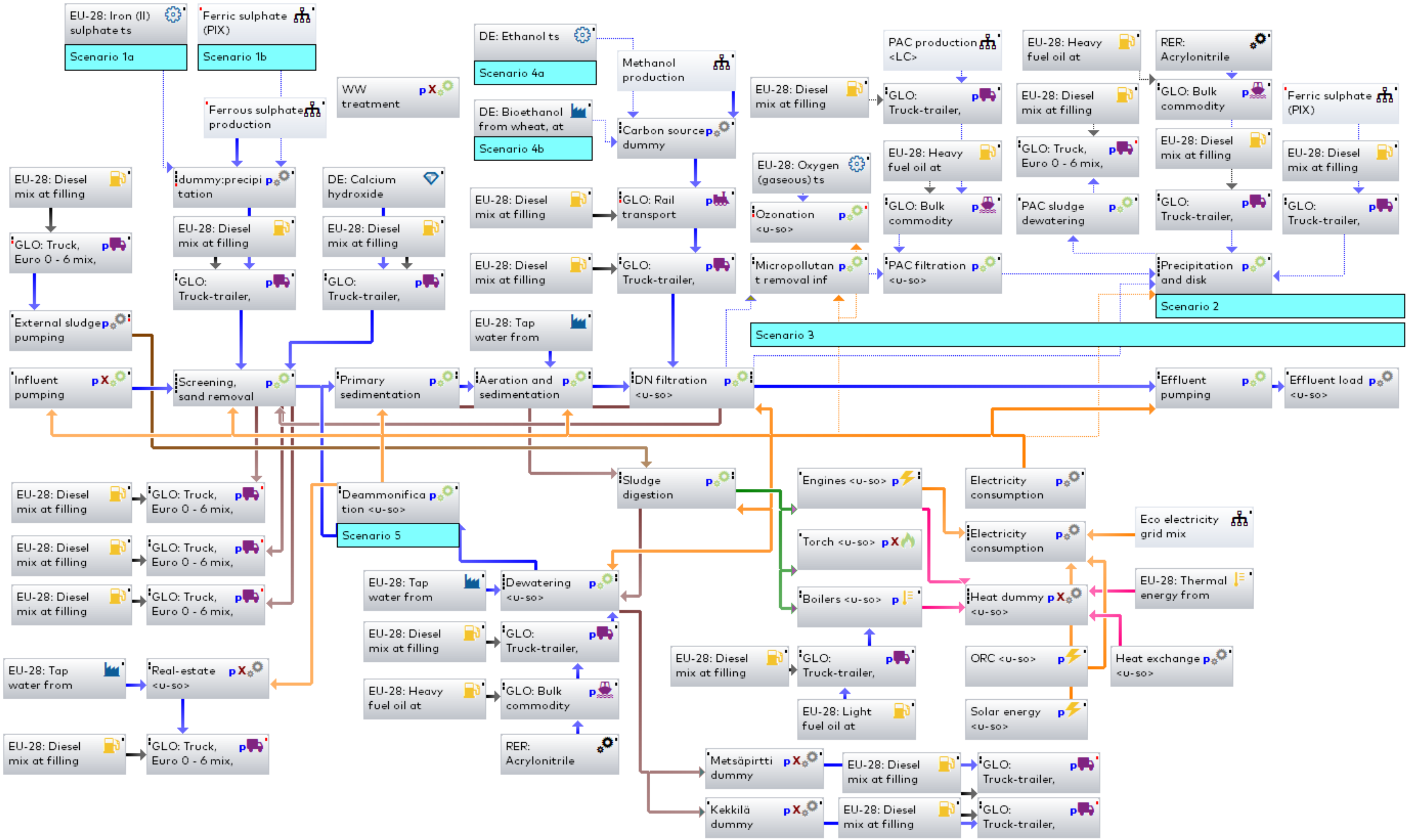

Figure 7: Overview of the GaBi model of Viikinmäki process

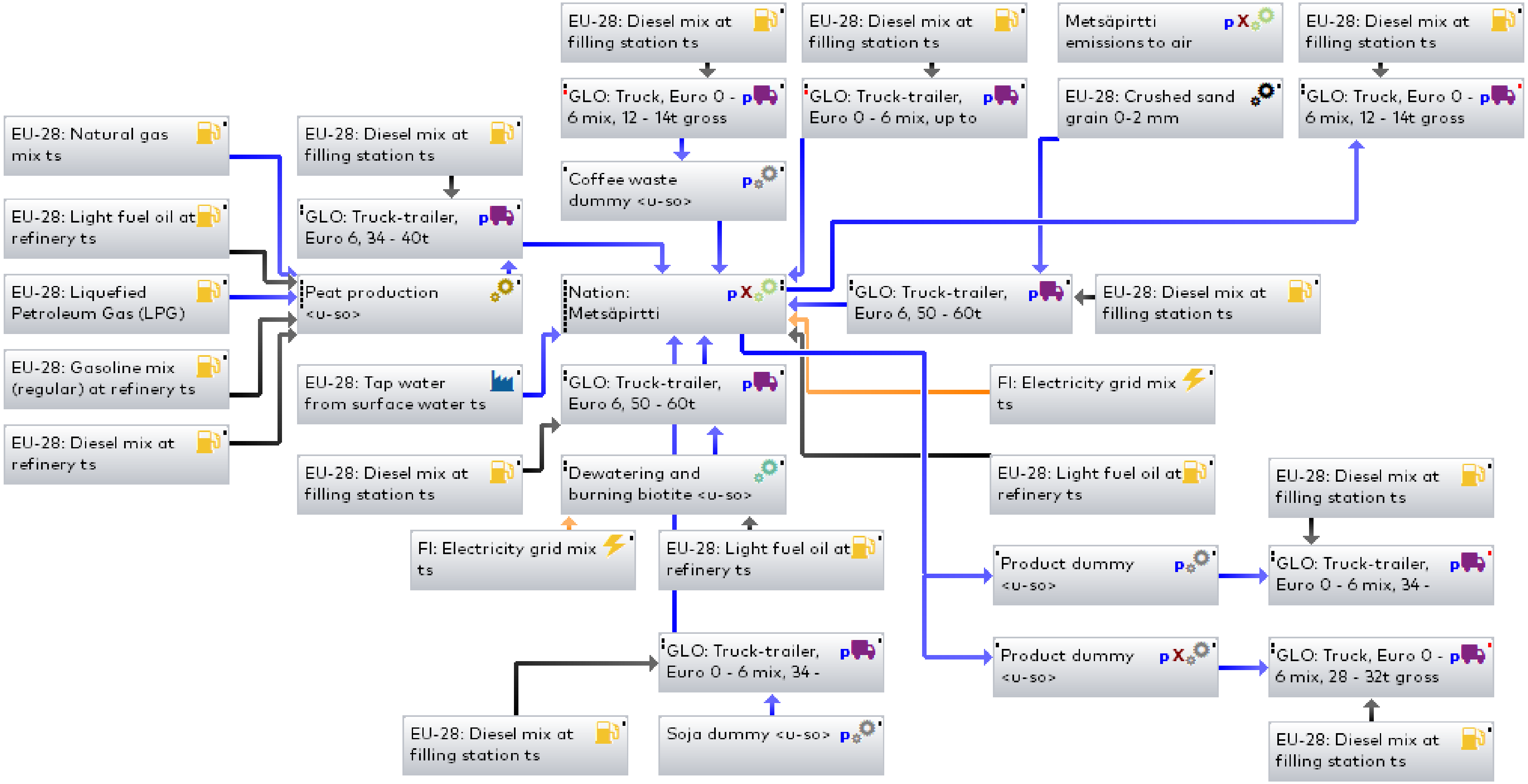

Figure 8: Overview of the GaBi model of Metsäpirtti process

7.2 Total Results for the Current Processes

Results for Viikinmäki and Metsäpirtti processes were calculated separately for the current processes and each scenario in all impact categories listed earlier in ***Table 7***.

The overall results for the current processes of Viikinmäki and Metsäpirtti are presented in ***Table 9***. The results were also calculated in values for one PE (***Table 10***) and 1000 m^3 of influent (***Table 11***). The total results are opened further in sections 0–7.7. Detailed results for different process factors are presented for the current Viikinmäki process in ***Appendix 8*** and for the Metsäpirtti process in ***Appendix 9***. The Metsäpirtti results include sludge handling from both Viikinmäki and Suomenoja WWTPs, 90% of which came from Viikinmäki.

Table 9: Total results for Viikinmäki and Metsäpirtti processes in 2018

Impact category	Abbreviation	Viikinmäki result	Metsäpirtti result	Unit
Abiotic depletion potential	ADP elements	0.95	0.35	kg Sb eq.
Abiotic depletion potential of fossil resources	ADP fossil	1.24×10^8	$1,94\times10^8$	MJ
Acidification potential	AP	4.35×10^4	$3,25\times10^4$	kg SO2 eq.
Eutrophication potential	EP	3.57×10^5	$2,49\times10^4$	kg PO4 eq.
Freshwater aquatic ecotoxicity potential	FAETP	8.74×10^4	$1,35\times10^4$	kg DCB eq.
Global warming potential*	GWP	3.84×10^7	$2,15\times10^7$	kg CO_2 eq.
Human toxicity potential	HTP	1.05×10^5	$1,37\times10^5$	kg DCB eq.
Marine aquatic ecotoxicity potential	MAETP	6.94×10^7	$7,58\times10^7$	kg DCB eq.
Ozone layer depletion potential	ODP	0.85	$1,01\times10^{-8}$	kg R11 eq.
Photochemical ozone creation potential	POCP	3.98×10^3	$9,65\times10^2$	kg C_2H_4 eq.
Terrestric ecotoxicity potential	TETP	7.15×10^3	$3,6\times10^3$	kg DCB eq.

*100 years, excluding biogenic carbon

Table 10: The total LCIA results for one PE

Impact category	Viikinmäki result	Metsäpirtti result	Unit
ADP elements	8.65×10^{-7}	3.19×10^{-7}	kg Sb eq./PE
ADP fossil	113	176	MJ/PE
AP	0.04	0.03	kg SO2 eq. /PE
EP	0.32	0.02	kg PO4 eq. /PE
FAETP	0.08	0.01	kg DCB eq. /PE
GWP*	34.9	19.5	kg CO2 eq. /PE
HTP	0.10	0.12	kg DCB eq. /PE
MAETP	63.1	68.9	kg DCB eq. /PE
ODP	7.73×10^{-7}	9.16×10^{-15}	kg R11 eq. /PE
POCP	3.62×10^{-3}	8.77×10^{-4}	kg C_2H_4 eq. /PE
TETP	6.50×10^{-3}	3.34×10^{-3}	kg DCB eq. /PE

*100 years, excluding biogenic carbon

Table 11: The total LCIA results for 1000 m^3 of influent

Impact category	Viikinmäki result	Metsäpirtti result	Unit
ADP elements	1.03×10^{-5}	3.80×10^{-6}	kg Sb eq. / 1000 m^3
ADP fossil	1.34×10^{3}	2.10×10^{3}	MJ / 1000 m^3
AP	0.47	0.35	kg SO2 eq. / 1000 m^3
EP	3.86	0.26	kg PO4 eq. / 1000 m^3
FAETP	0.95	0.15	kg DCB eq. / 1000 m^3
GWP*	416	232	kg CO2 eq. / 1000 m^3
HTP	1.14	1.48	kg DCB eq. / 1000 m^3
MAETP	751	820	kg DCB eq. / 1000 m^3
ODP	9.20×10^{-6}	1.09×10^{-13}	kg R11 eq. / 1000 m^3
POCP	0.04	0.01	kg C_2H_4 eq. / 1000 m^3
TETP	0.08	0.04	kg DCB eq. / 1000 m^3

*100 years, excluding biogenic carbon

7.3 Results by Categories for the Current Viikinmäki Process

For further examination of the results, the Viikinmäki process was divided into seven life cycle categories (***Table 12***).

Table 12: Constitution of different life cycle categories of Viikinmäki process

Life cycle category	Factors causing impact potential in the category
WWT	Direct gaseous emissions from the process
Sludge treatment	Direct gaseous emissions from the process
Chemicals	Production of process chemicals
Transports	All transports of chemicals, waste and sludge to and from the plant
Real-estate	Water used in the real-estate (excluding the treatment process)
Energy	Production of all purchased energy and on-site production
Recipient	Effluent load to the sea

Figure 9 shows the contribution of different life cycle categories to LCIA total results of the current Viikinmäki process. Positive shares in the figure indicate that the category increased the potential environmental impact and negative shares again caused a counter impact, decreasing the total result. Impact categories ADP (elements) and ODP had such low overall values that they were considered insignificant to the total environmental impact and were thus excluded from further analysis. The total results of the different categories are of different scales and they are presented on the right. When interpreting the results from the figures in this chapter, it should be noted that the different impact category results cannot be directly compared with each other due to different units and the types of impact. The results are further opened with the different life cycle categories.

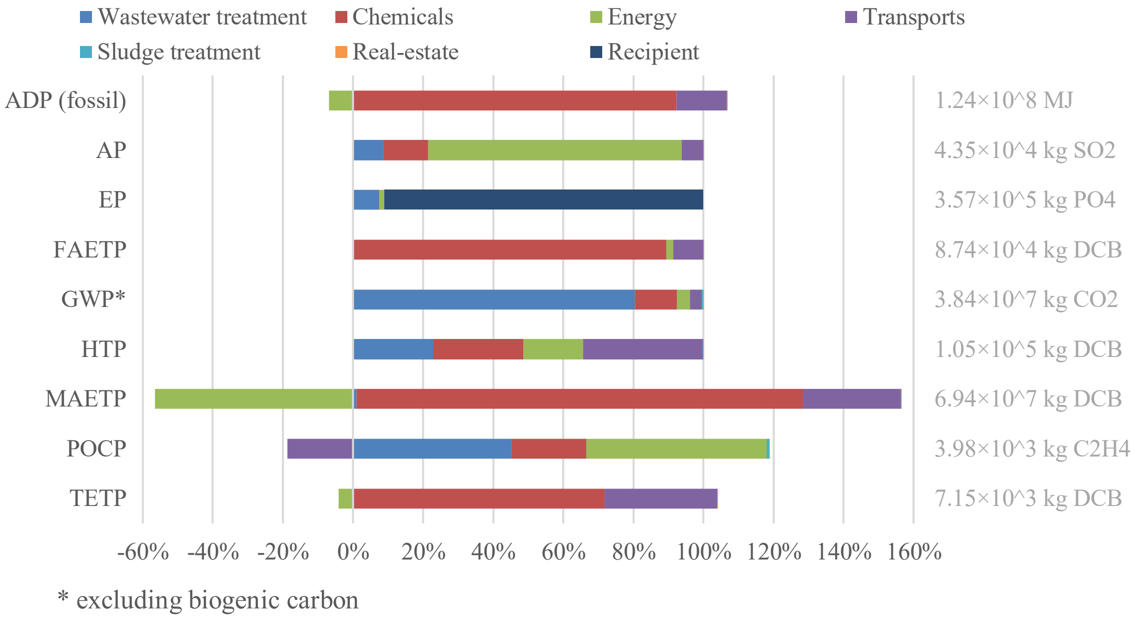

Figure 9: Shares of contributing categories to LCIA total results of the current Viikinmäki WWT process.

It can be noted from ***Figure 9*** that chemicals yielded the biggest percentage of the total result, causing most of the impact potential in ADP (fossil) category and toxicity-related categories. ***Figure 10*** shows contributions of different chemicals to these impact categories. The total bars present the full result of the impact category and the contribution of different chemicals is presented with the colored sections. It can be noted that methanol was the biggest cause of environmental impact potential in most impact categories. The impacts of methanol production were mainly related to fossil fuel consumption in ADP (fossil) and in other impact categories the effect came from emissions to air and emissions to fresh waters from the production process.

Also, calcium hydroxide and polymer caused relevant shares of chemical impacts considering all categories, calcium hydroxide being more dominant in the ADP (fossil) and toxicity-related categories with most impact from chemical production. Calcium hydroxide had the biggest impact share from chemicals in TETP causing nearly 98% of the impact of all chemicals and 70% of the entire environmental impact category. Impacts from both chemicals mainly originated from emissions to air from the production processes. Ferrous sulphate caused a very minor impact compared to the other chemicals as it was made from a byproduct and therefore didn't consume a lot of raw materials. Also, it required less processing steps than a chemical produced from raw materials.

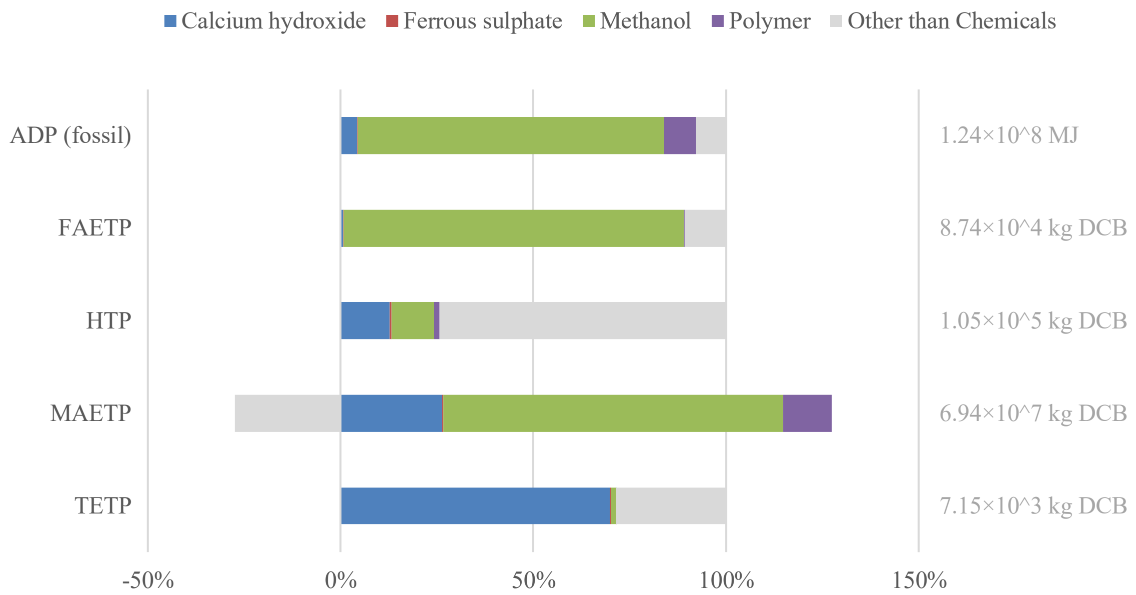

Figure 10: Impact categories where chemical production causes relevant effect: Split into process parts

The WWT process was the main contributor of GWP and it also had notable impacts on POCP and HTP. The division of WWT impacts in these categories is presented in ***Figure 11***. The impact is mainly caused by two GHGs, namely nitrous oxide and methane. Nitrous oxide produced 80% of the impact from WWT and 65% of the total Viikinmäki GWP. Methane again had the biggest contribution in POCP with a 73% share of WWT POC and 33% share of total POC. In HTP, the main cause of impact from WWT were the benzene emissions from the process with a 20% share of the Viikinmäki overall HTP.

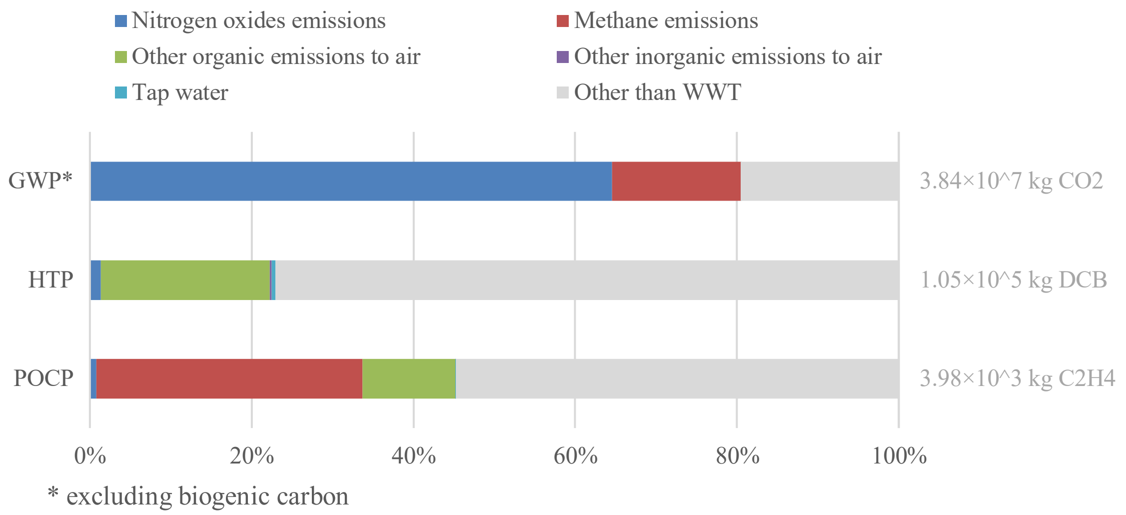

Figure 11: Impact categories where the WWT process causes relevant effect: Split into process parts

Energy caused a relevant share of the positive impact in POCP, HTP and AP. It also caused a negative impact in and MAETP and HTP. Division of factors causing energy impacts is presented in ***Figure 12***. Most impact was caused by engines, more precisely their nitrogen and sulphur oxides emissions. Engines produce electricity for the WWT process, thus the impact of the category could be allocated to different process parts. Half of the electricity produced was used in aeration. The next biggest electricity consumers were influent pumping and real-estate with 15% and 13% shares respectively.

A negative impact decreases the total potential impact by for example saving resources or lowering emissions. Energy caused quite a substantial negative impact on MAETP and small negative impacts also on TETP and ADP (fossil). In HTP, the positive and negative impacts ruled out each other making the overall result from energy near zero. Negative impact from energy category was caused by extra heat production sold to the energy market which is assumed to decrease the need for heat production from e.g. fossil fuels.

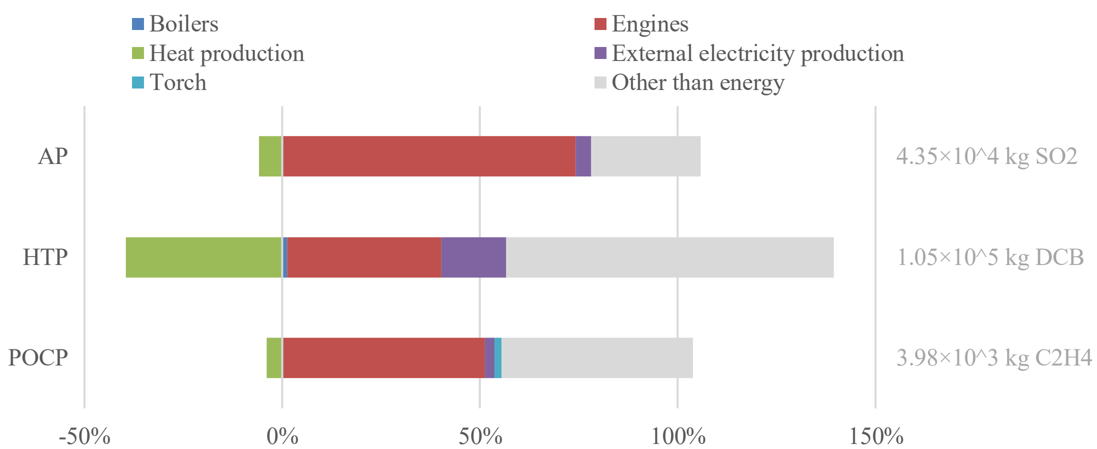

Figure 12: Impact categories where energy causes relevant effect: Split into process parts

Transports caused the most relevant shares of positive impact in ADP (fossil), HTP, MAETP and TETP and a negative impact in POCP. Shares of contribution can be seen in ***Figure 13***. The impacts from transports of different materials followed roughly the ratio of transported masses, thus clearly the biggest contributor to transport impacts were external sludge transports to the plant. In ADP (fossil), MAETP and TETP the positive impact was caused by fuel production but in HTP also the emissions from the transport process itself increased the impact potential.

According to GaBi results, transports had a negative POCP. This resulted from nitrogen monoxide emissions which have a negative impact factor value in the CML method. According to Thinkstep (2019c), this negative factor is due to nitrogen monoxide's ability to transform ozone back into oxygen. However, this impact is local and affected by weather conditions which makes it uncertain.

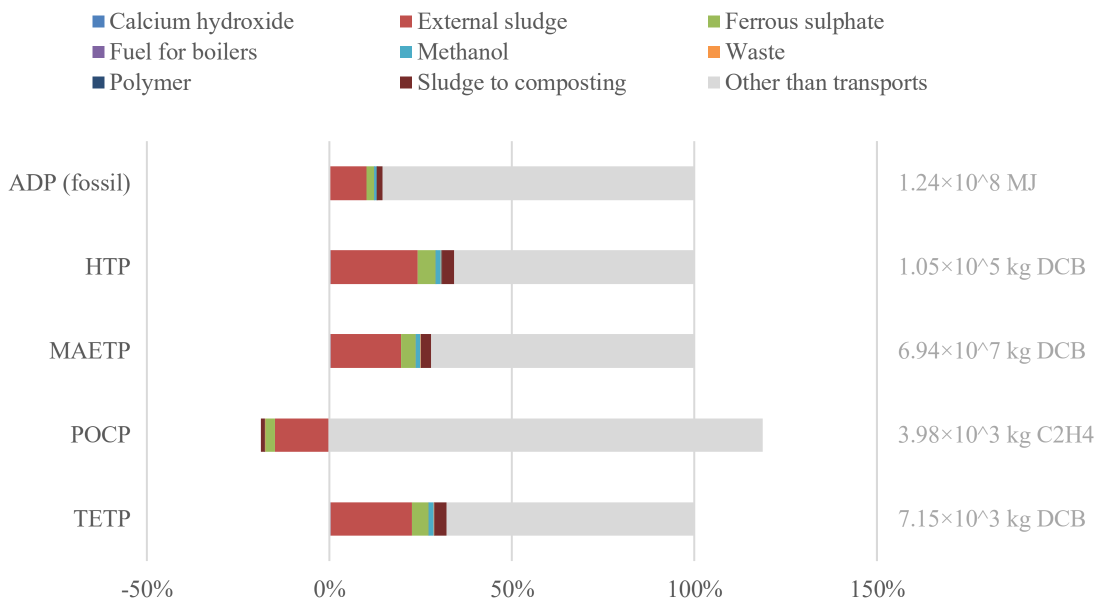

Figure 13: Impact categories where transports cause relevant effect: Split into process parts

The effluent load to the sea caused the clear majority of EP. In other categories it caused a zero impact. Nitrogen was the most relevant nutrient in the effluent covering nearly 55% of the total potential caused by effluent load (***Figure 14***). The next biggest contributors in the category were COD load and phosphorus load with 25% and 13% shares respectively.

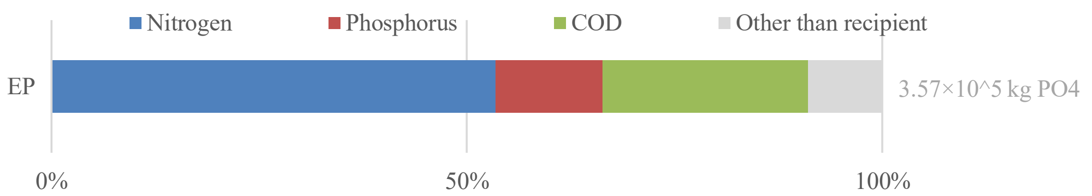

Figure 14: Contribution of effluent load in EP: Split between different emissions

Sludge treatment steps at the WWTP and real-estate did not cause a notable share of impact in any impact category.

7.4 Results by Categories for the Metsäpirtti Process

As for the Viikinmäki process, the Metsäpirtti process was divided into life cycle categories for further examination of results (***Table 13***).

Table 13: Constitution of different life cycle categories of Metsäpirtti process

Life cycle category	Factors causing impact potential in the category
Composting and soil production process	Direct gaseous emissions from the process and water consumption in the process
Additives	Production of substances added in compost and soil
Transports	All transports of additives to the facility and waste and products from the plant
Energy	Production of electricity used at the facility and fuel for the machinery

Figure 15 shows the contribution of different life cycle categories to LCIA total results of the current Metsäpirtti composting and soil production process. As in results from Viikinmäki, ADP (elements) and ODP had very low overall values and were excluded from further analysis.

It can be seen from ***Figure 15*** that impacts from Metsäpirtti process were mainly derived from the process itself, production of additives and transports. Energy produced quite low impacts.

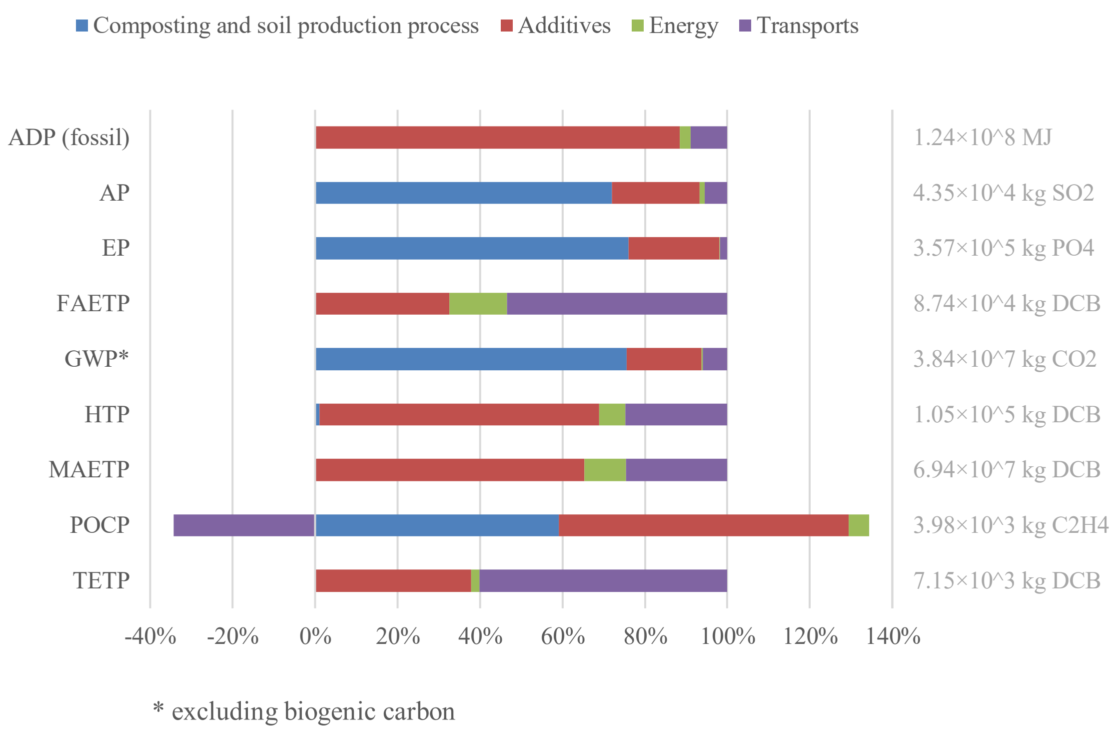

Figure 15: Shares of contributing categories to LCIA total results of the Metsäpirtti composting and soil production process

Production of compost and soil additives had a high impact on the results in all categories. Ratios of different additives contributing to the results can be seen in ***Figure 16***. Nearly all the impact potentials caused by additional substances production came from peat and sand. Peat was the biggest singular contributor for Metsäpirtti total ADP (fossil) and POCP with 79% and 59% shares respectively. Also, sand had the highest total effect on MAETP and HTP with 47% and 50% respectively. Biotite had nearly no contribution to the total impact potentials.

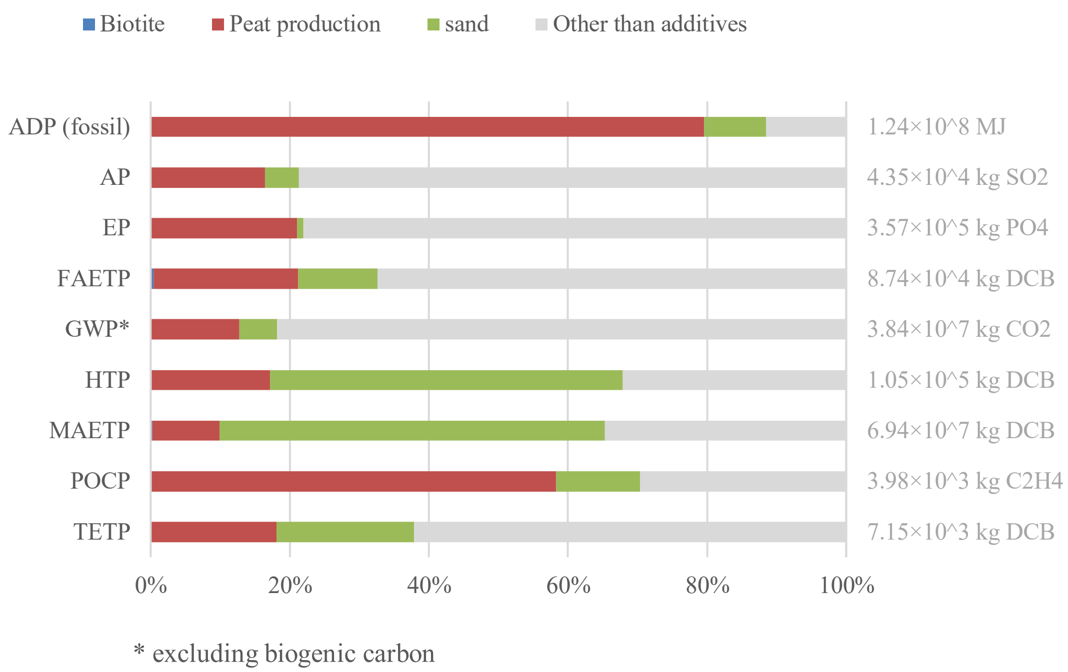

Figure 16:Impact categories where additives cause relevant effect on Metsäpirtti results: Split into process parts

Contribution of the composting and soil production process itself is presented in ***Figure 17***. Nitrous oxide emissions from the process increased EP and GWP the most. Gaseous ammonia emissions to air caused the biggest increase in AP and methane in POCP. These emissions were also the main contributors to these four impact categories in Metsäpirtti total results.

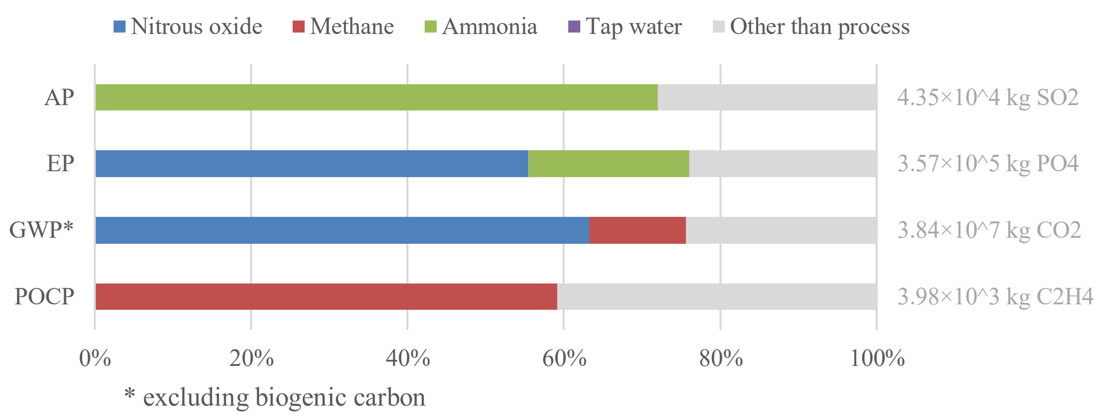

Figure 17:Impact categories where composting and soil production process causes relevant effect on Metsäpirtti results: Split into process parts

Division of transports in the most relevant impact categories is presented in ***Figure 18***. Transports with the highest masses produced the biggest impact potentials, thus product transports were the biggest contributor in all impact categories presented. Impacts from additives transports were lower due to lower mass and better European emission standards of the vehicles. As in Viikinmäki results, transports caused a negative effect on POCP where again products had the biggest negative share.

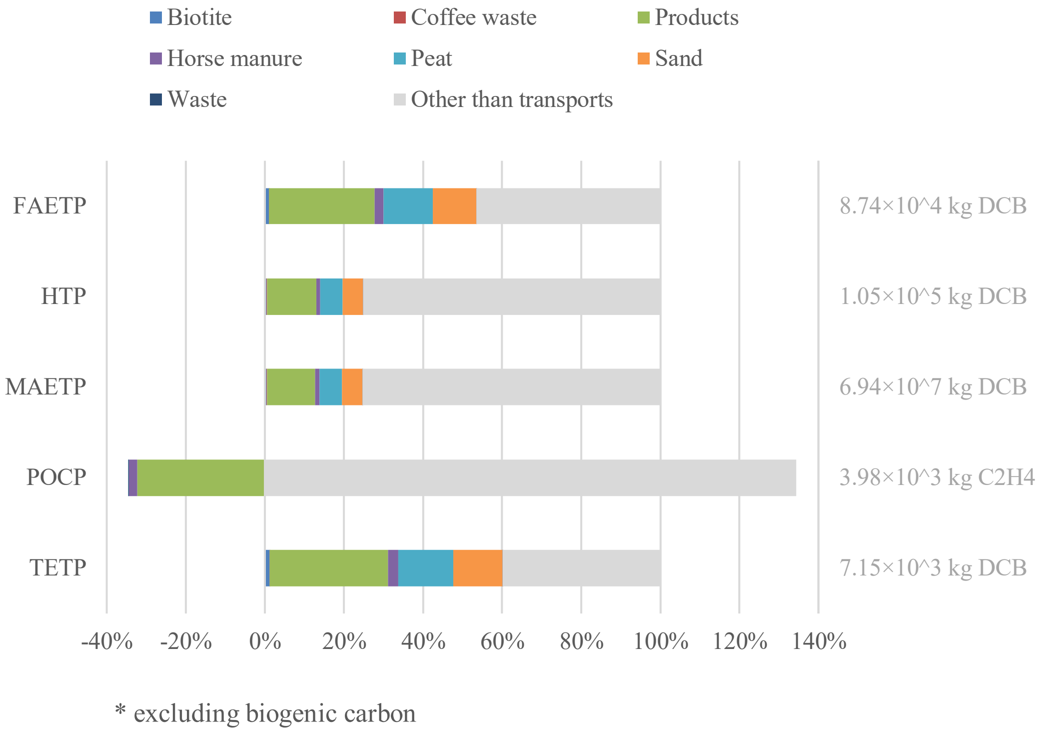

Figure 18:Impact categories where transports cause relevant effect on Metsäpirtti results: Split into process parts

Energy had a low impact in all categories. The highest contribution from energy was 14% in FAETP where the cause was mainly light fuel oil used as fuel of the machinery.

7.5 LCIA Results from different scenarios

The total results for studied scenarios in different impact categories are first presented in ***Table 14*** and their percentages compared to the initial result in ***Table 15***. The results are then analyzed further for each scenario. The numeric changes to the total results caused by different factors are presented in ***Appendix 10.***

Table 14: Total results for different scenarios

Scenario	ADP elements (kg Sb eq.)	ADP fossil (MJ)	AP (kg SO2 eq.)	EP (kg PO4 eq.)	FAETP (kg DCB eq.)	GWP* (kg CO2 eq.)	HTP (kg DCB eq.)	MAETP (kg DCB eq.)	ODP (kg R11 eq.)	POCP (kg C_2H_4 eq.	TETP (kg DCB eq.)
Viikinmäki process	0,95	$1{,}24\times10^{8}$	$4{,}35\times10^{4}$	$3{,}57\times10^{5}$	$8{,}74\times10^{4}$	$3{,}84\times10^{7}$	$1{,}05\times10^{5}$	$6{,}94\times10^{7}$	0,85	$3{,}98\times10^{3}$	$7{,}15\times10^{3}$
1a	1,51	$1{,}95\times10^{8}$	$8{,}38\times10^{4}$	$3{,}57\times10^{5}$	$1{,}13\times10^{5}$	$4{,}02\times10^{7}$	$3{,}03\times10^{5}$	$2{,}19\times10^{8}$	0,85	$6{,}12\times10^{3}$	$9{,}48\times10^{3}$
1b	1,02	$1{,}37\times10^{8}$	$5{,}14\times10^{4}$	$3{,}57\times10^{5}$	$9{,}20\times10^{4}$	$3{,}89\times10^{7}$	$1{,}46\times10^{5}$	$1{,}29\times10^{8}$	0,85	$4{,}41\times10^{3}$	$7{,}59\times10^{3}$
2	1,43	$1{,}34\times10^{8}$	$4{,}66\times10^{4}$	$3{,}41\times10^{5}$	$8{,}93\times10^{4}$	$3{,}89\times10^{7}$	$1{,}21\times10^{5}$	$9{,}57\times10^{7}$	0,85	$4{,}27\times10^{3}$	$7{,}42\times10^{3}$
3	5,84	$3{,}24\times10^{8}$	$9{,}60\times10^{4}$	$3{,}47\times10^{5}$	$1{,}22\times10^{5}$	$5{,}34\times10^{7}$	$9{,}59\times10^{5}$	$1{,}48\times10^{9}$	0,85	$7{,}31\times10^{3}$	$1{,}84\times10^{4}$
4	2,41	$4{,}92\times10^{7}$	$4{,}59\times10^{4}$	$3{,}62\times10^{5}$	$5{,}01\times10^{4}$	$3{,}80\times10^{7}$	$2{,}15\times10^{5}$	$6{,}43\times10^{7}$	0,85	$3{,}53\times10^{3}$	$8{,}31\times10^{3}$
5a	0,93	$1{,}10\times10^{8}$	$4{,}29\times10^{4}$	$3{,}57\times10^{5}$	$7{,}78\times10^{4}$	$3{,}76\times10^{7}$	$9{,}99\times10^{4}$	$5{,}69\times10^{7}$	0,85	$3{,}91\times10^{3}$	$5{,}78\times10^{3}$
5b	0,93	$1{,}10\times10^{8}$	$4{,}29\times10^{4}$	$3{,}60\times10^{5}$	$7{,}78\times10^{4}$	$4{,}14\times10^{7}$	$9{,}99\times10^{4}$	$5{,}69\times10^{7}$	0,85	$3{,}91\times10^{3}$	$5{,}78\times10^{3}$
5c	0,93	$1{,}10\times10^{8}$	$4{,}29\times10^{4}$	$3{,}55\times10^{5}$	$7{,}78\times10^{4}$	$3{,}58\times10^{7}$	$9{,}99\times10^{4}$	$5{,}69\times10^{7}$	0,85	$3{,}91\times10^{3}$	$5{,}78\times10^{3}$

*excluding biogenic carbon

Table 15: Shares of scenario results compared to the current process

Scenario	ADP elements (%)	ADP fossil (%)	AP (%)	EP (%)	FAETP (%)	GWP* (%)	HTP (%)	MAETP (%)	ODP (%)	POCP (%)	TETP (%)
1a	159	157	193	100	129	105	288	315	100	154	133
1b	107	111	118	100	105	101	138	186	100	111	106
2	151	108	107	96	102	101	115	138	100	107	104
3	615	261	221	97	140	139	911	2126	100	184	258
4	254	40	106	101	57	99	205	93	100	89	116
5a	98	89	99	100	89	98	95	82	100	98	81
5b	98	89	99	101	89	108	95	82	100	98	81
5c	98	89	99	99	89	93	95	82	100	98	81

*excluding biogenic carbon

In scenarios 1a and 1b (***Figure 19***) changing the precipitation chemical to ferrous sulphate manufactured via synthesis route clearly increased MAETP, HTP, AP and ADP. Using ferric sulphate made from byproduct ferrous sulphate increased MAETP and HTP but the effect was lower than with synthesis of ferrous sulphate. The increase in the impact potential was mainly due to more processing steps in the production and not due to increased consumption of coagulant to achieve the same treatment result.

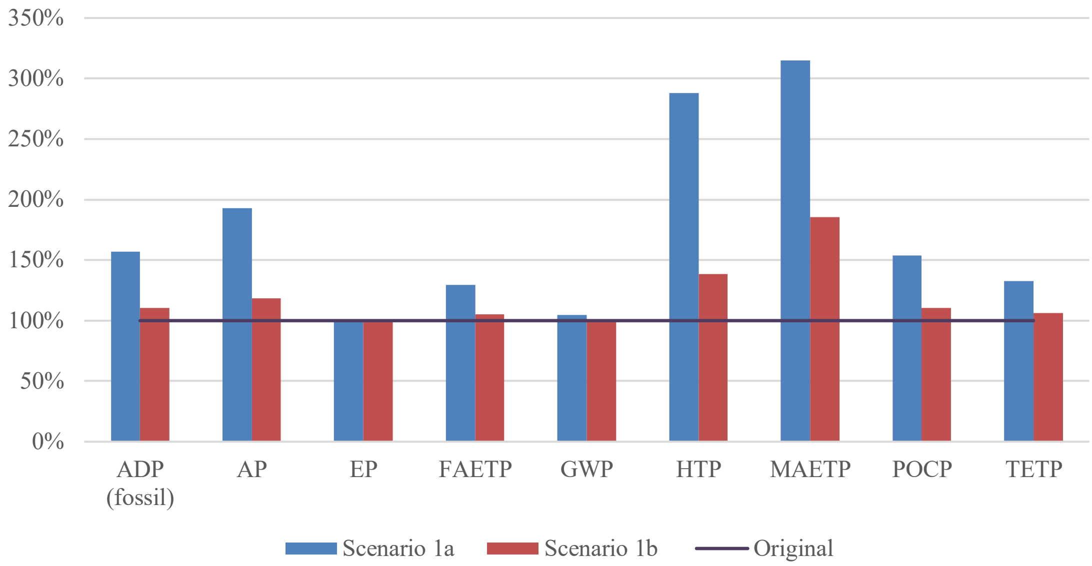

Figure 19: Changes to impact potentials from scenarios 1a (using non-byproduct-based ferrous sulphate) and 1b (using ferric sulphate)

In scenario 2, precipitation and disc filtration were added before effluent pumping which increased the consumption of chemicals and electricity but lowered the amount of phosphorus in the effluent. Again, mainly MAETP was increased to around 138% of the original situation (***Figure 20***). This was due to combined effect from ferric sulphate and polymer production and higher electricity consumption. Ferric sulphate and polymer are needed for precipitation and floc forming. Increase in electricity consumption is mostly related to backwashing of the filter (Langer & Schermann 2013, p. 58).

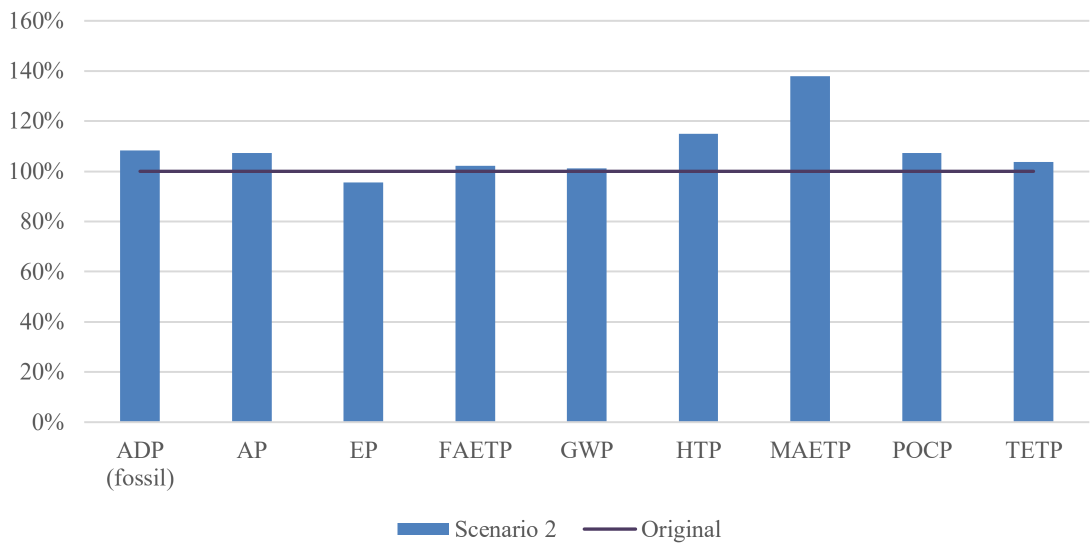

Figure 20: Changes to impact potentials from scenario 2 (adding effluent polishing)

In scenario 3 (***Figure 21***), the removal of micropollutants had the biggest effect on all other impact categories except for EP. MAETP was over 22-fold and HTP 9-fold compared to the original situation. The effects were mainly due to PAC manufacturing from fossil fuels which contributed around 60% of the total impact potential in both categories. Another addition increasing the impact potentials was oxygen production for ozonation which had a roughly 15% share in both categories.

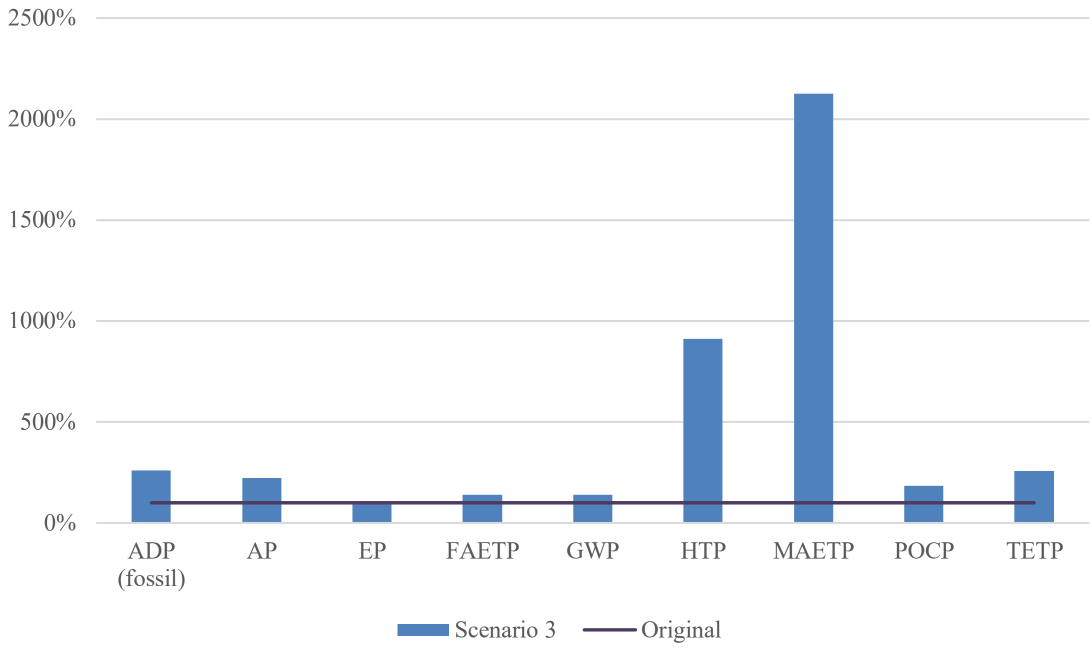

Figure 21:Changes to impact potentials from scenario 3 (adding micropollutant removal)

Changing the carbon source of DN filtration from fossil fuel-based methanol to bio-based ethanol decreased ADP (fossil) and FAETP (***Figure 22***) due to less fossil fuels consumption and emissions to fresh waters from manufacturing. Also, a lower dose of ethanol due to higher COD compared to methanol, caused some decrease. However, HTP was increased with higher emissions from the production process.

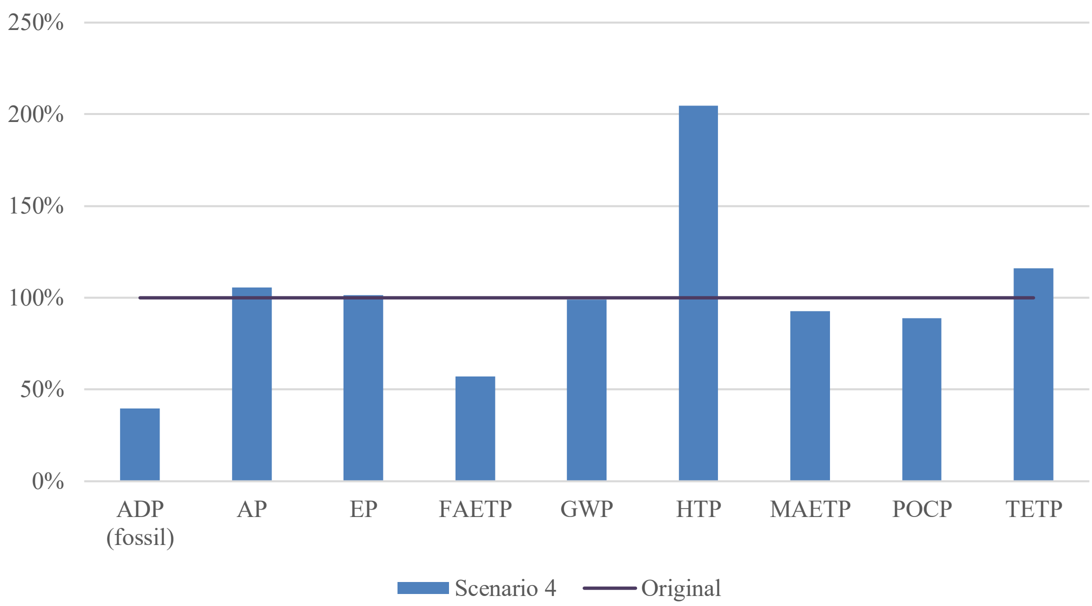

Figure 22: Changes to impact potentials from scenario 4 (using bio-ethanol)

As presented in ***Figure 23***, deammonification of all reject water had mainly decreasing effects to different impact potentials with all three N_2O emission sub-scenarios. Only GWP increased with roughly 10% in scenario 5b due to higher N_2O emissions.

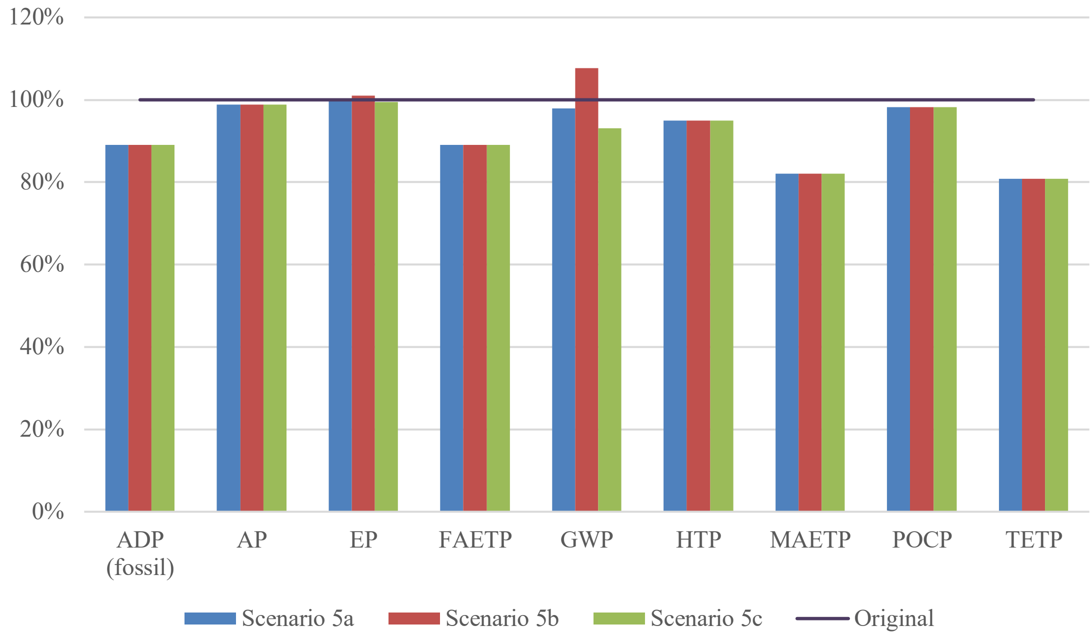

Figure 23: Changes to impact potentials from scenarios 5a (no changes in emissions), 5b (higher emissions) and 5c (lower emissions) treating reject water.

7.6 Normalization of LCIA Results

Results from the current processes LCIA were normalized by placing them into European context to estimate which impact categories were the most significant for the Viikinmäki and Metsäpirtti processes. The used method for normalization was CML 2001 – April 2015, EU25+3, year 2000 excluding biogenic carbon. The results were the shares of Viikinmäki and Metsäpirtti environmental impact potentials compared to the sum values of EU countries for the impact potentials. Normalization of the current process results for Viikinmäki and Metsäpirtti are presented in ***Figure 24***. Numerical values are collected in ***Appendix 11.***

According to normalization, the impact categories affected most by Viikinmäki and Metsäpirtti current processes were EP, GWP and ADP (fossil). The category with the highest result, EP, consisted mostly of effluent nitrogen load. High GWP result was mainly due to N_2O emissions from both Viikinmäki and Metsäpirtti processes. The biggest factor increasing ADP (fossil) for Viikinmäki was methanol production while for Metsäpirtti it was peat production. It can be stated that these process variables had the biggest environmental impacts when considered in broader context.

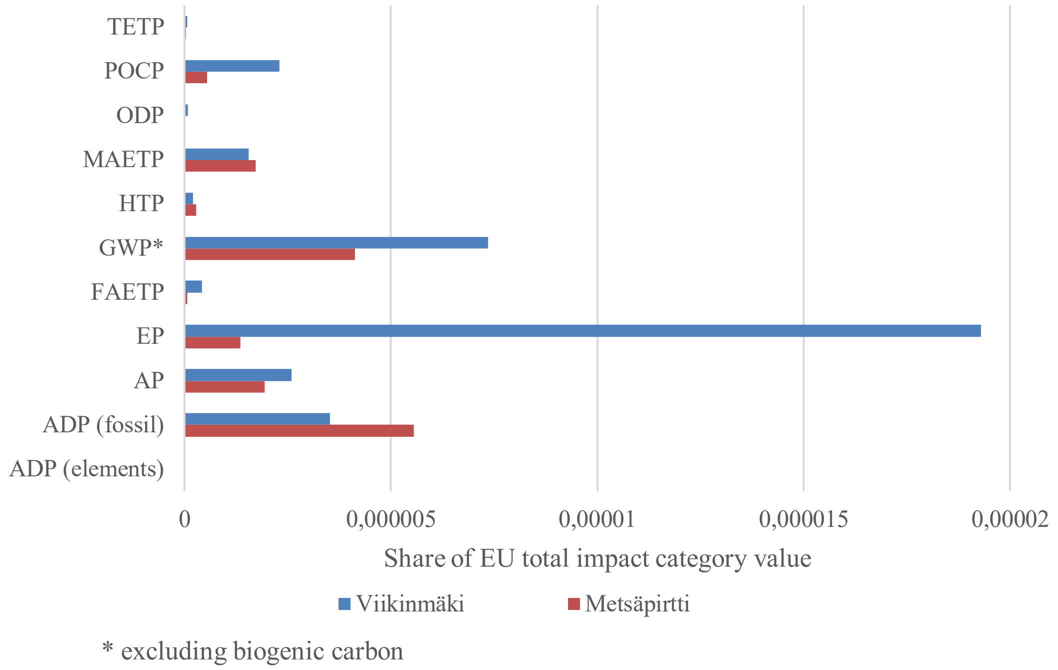

Figure 24: Normalized results of LCIA for the current Viikinmäki and Metsäpirtti processes

The normalization of results was also conducted for scenario 3 as it caused a major rise in impact potentials in several categories. The normalized results of scenario 3 together with the current Viikinmäki process are presented in ***Figure 25.*** The numerical values are included in ***Appendix 11*** together with the current process results.

In scenario 3, PAC production raised MAETP above other categories in importance on EU-level. PAC was hence the overall biggest factor of the entire system causing environmental impacts.

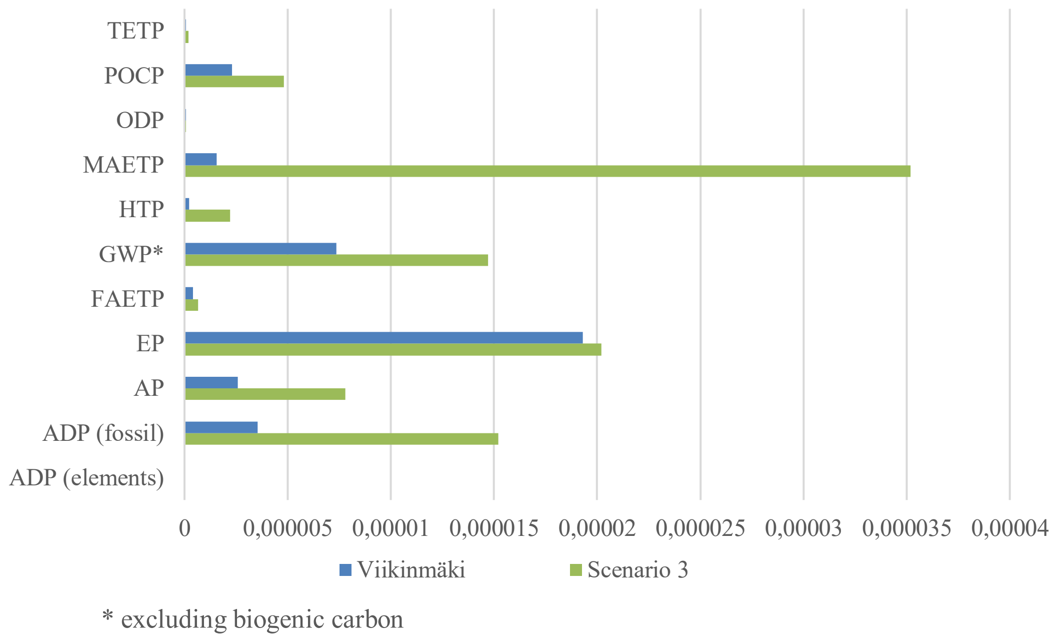

Figure 25: Normalized results for the current Viikinmäki process and scenario 3

7.7 Sensitivity Analysis

The sensitivity analysis for the LCA results was performed using the six biggest factors contributing to the environmental impact of both Viikinmäki and Metsäpirtti processes. These factors were nitrogen load to the sea from Viikinmäki, nitrous oxide emissions to air from Viikinmäki, nitrous oxide emissions to air from Metsäpirtti, methanol production, peat production and sand production. The studied impact categories were EP, GWP and ADP (fossil) as they had the highest normalized impacts.

Sensitivity was examined with both local parameter sensitivity analysis and Monte Carlo analysis. 5% standard deviation was used in the local parameter sensitivity analysis for all parameters changed. The results (***Appendix 12***) showed low sensitivity of results with the highest variation of -2,5% and +2,5% in EP due to nitrogen load in Viikinmäki effluent.

Monte Carlo analysis with 59 simulation runs was also executed using 5% standard deviation for the parameters listed above. Numerical results of the Monte Carlo analysis are listed in ***Appendix 13***. ***Figure 26*** shows division of simulation results in all three impact categories studied. Results showed little dispersion from the initial results.

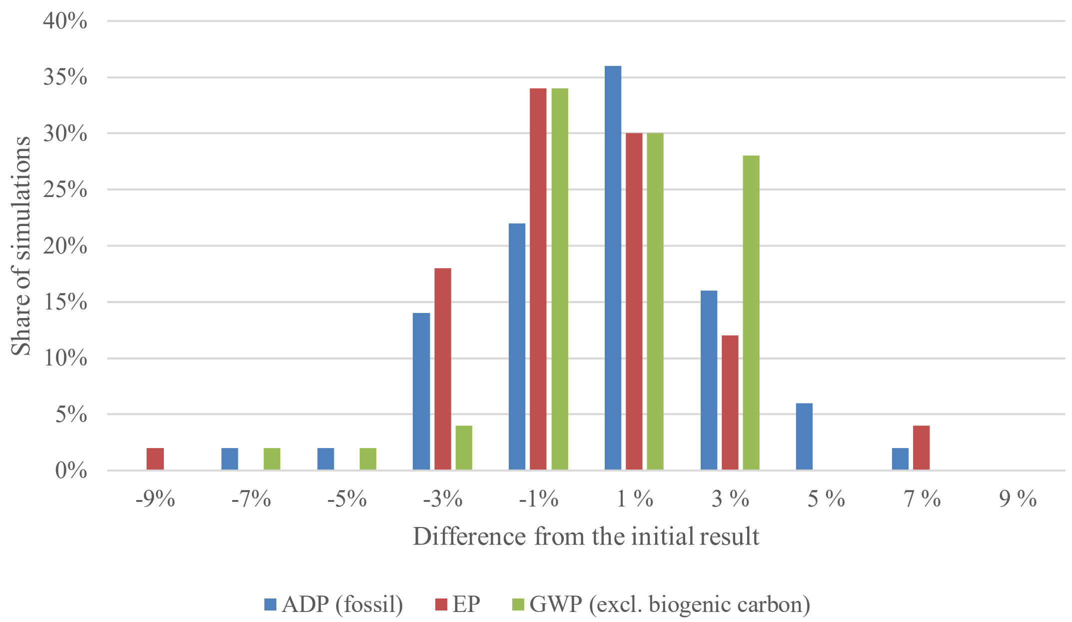

Figure 26: Results of Monte Carlo analysis

As EP was the most relevant category in the Viikinmäki total results, it was also calculated with another method, TRACI 2.1, which was designed for the North American context. The total result for EP for the Viikinmäki process was 3.2×10^5 kg N eq. TRACI 2.1 provides the result for EP in kilograms of nitrogen equivalent which is different to the unit of phosphate equivalent in CML. The CML result was converted to the same unit using a factor of 2,38 kg N / kg PO_4 from TRACI. The initial result in nitrogen equivalent was then 3.57×10^5 kg N eq. Thus, the results from CML and TRACI methods for EP were of the same magnitude which also indicates reliability of the EP result.

8 Carbon Footprint Calculation

In this chapter the results of the separate carbon footprint calculation are presented and compared with the GWP results from the full LCA. Sensitivity of the results is also analyzed. The separate carbon footprint was calculated with a Carbon Footprint Calculation Tool which is an Excel base for calculating carbon footprints of WWTPs. It was created by Gustavsson & Tumlin (2013) in their project for Swedish Water and it is freely downloadable from the website of VA-teknik Södra (2019).

8.1 Results from the Carbon Footprint Calculation Tool for Viikinmäki Process

Using the Carbon Footprint Calculation Tool, the total carbon footprint of year 2018 for the Viikinmäki process was approximately 43 600 tons of CO_{2e}. Divided by the amount of PEs treated and volume of influent the result was 41 kg CO_{2e}/PE/year and 472 kg CO_{2e}/ 1000 m3 /year respectfully. This was around 14% more than the GWP result from GaBi. Division of the total carbon footprint results together with GWP results is presented in ***Figure 27***. The numeric values of results are collected in ***Appendix 14***. As in the GWP result, the footprint consisted mostly of direct N_2O emissions from the WWT process. N_2O emissions accounted for 64% of the total carbon footprint followed by methane emissions with a 17% share.

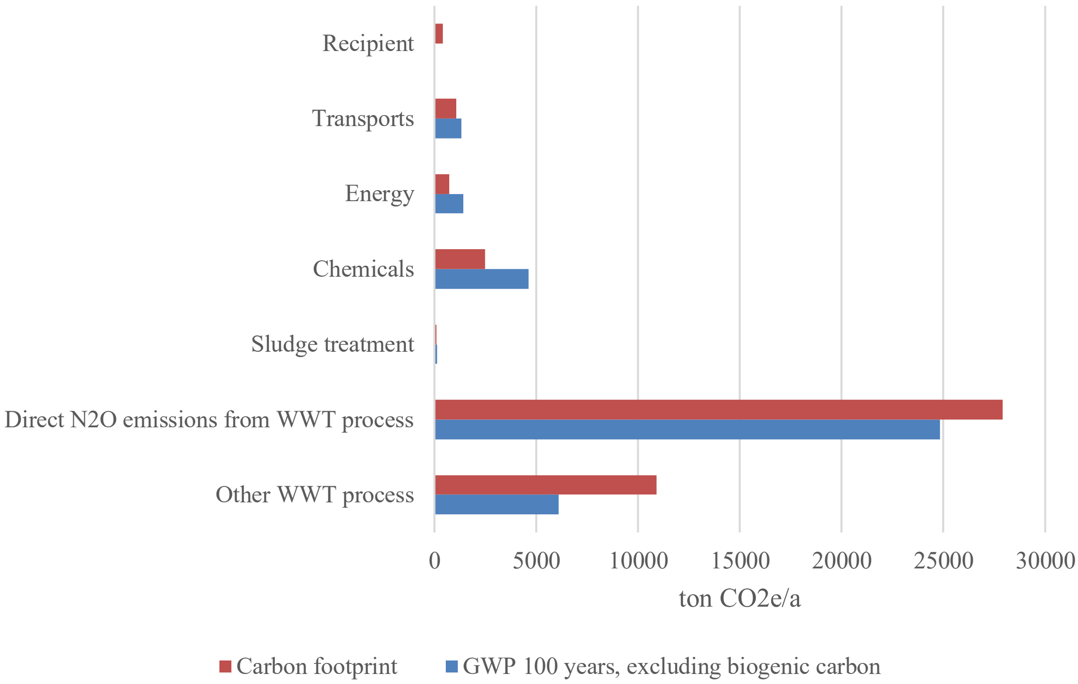

Figure 27: The Carbon Footprint Tool and GWP results for Viikinmäki, compared for different processes

The differences in results were mainly due to different constants used by the methods when turning the emissions into CO_2 equivalents. For example, the constant used in CML method was 265 for nitrous oxide and 28 for methane while in the Carbon Footprint Calculation Tool they were 298 for nitrous oxide and 34 for methane. The difference in the impact of chemicals was due to higher CO_2 equivalents for all chemicals in the CML method. The difference was the highest for calcium hydroxide: The impact of calcium hydroxide with the Carbon Footprint Calculation Tool was only 27% of the impact with CML method.

The differences in results from GaBi and the Carbon Footprint Calculation Tool indicate some sensitivity of the GWP and carbon footprint results when using different methods. However, the relationships between different contributing factors in the process stayed close to each other in both calculations.

8.2 Results from the Carbon Footprint Calculation Tool for Metsäpirtti Process

For the Metsäpirtti process, the calculated carbon footprint was 10 300 tons of CO_{2e} which was 52% less than the GWP result. Divided by the amount of PEs treated and volume of influent the result was 9.7 kg CO_{2e}/PE/year and 111 kg CO_{2e}/ 1000 m3 /year respectfully Precise division of carbon footprint results for Metsäpirtti is gathered in ***Appendix 15***. Division of carbon footprint and GWP results compared is presented in ***Figure 28***. The difference in results originated mainly from very different assumed direct emissions from the composting process. The carbon footprint of the direct emissions was only 11% of the GWP result. In the Carbon Footprint Calculation Tool, the emissions from composting were calculated with factors in the tool and no measured data was used. In the GaBi model, the emissions were estimated with measurement data from year 2011. Both estimations include many uncertainties.

The carbon footprint of composting and soil production was also lowered a bit by the fact that in the calculation, end use of compost and soil was considered which caused a negative impact. This, however only lowered the carbon footprint by less than 1%.

For other additives than peat, carbon footprints were assumed near zero as they were made mainly from byproducts. This was not the case in GWP where processing steps for the byproduct were considered and the result was thus higher. However, in the carbon footprint calculation, peat produced more than double result than in GWP. The difference originated from differences in the assumed production process of peat: The process was modelled in GaBi using inventory data from a study by Boldrin *et al.* (2010) while the Carbon Footprint Calculation Tool included an estimated emission factor for peat.

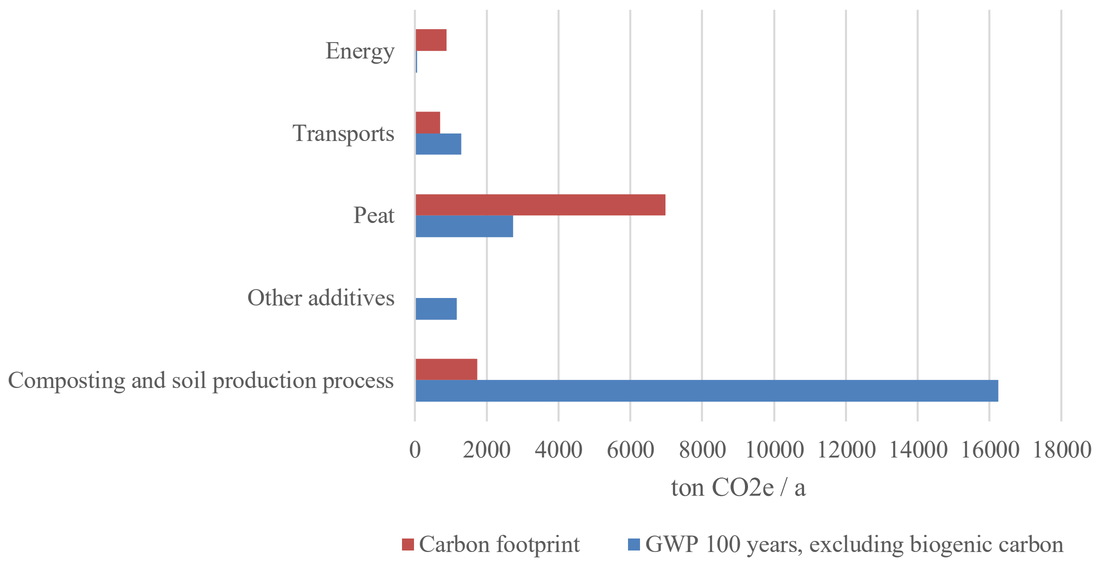

Figure 28: The Carbon Footprint Tool and GWP results for Metsäpirtti, compared for different processes

8.3 Sensitivity of Carbon Footprint Results

The emission factors used in the Carbon Footprint Calculation Tool were often averages of minimum and maximum values found from different written sources presented in the tool. In some cases, an average between calculated and measured values was used. Variation of the carbon footprint calculated was tested by applying these optional factors. Detailed descriptions of changed parameters and results are collected in ***Appendix 16***. ***Figure 29*** shows that with different utilized factors, the total carbon footprint varied greatly. Also, the GWP results from the full LCA are shown in the graph for comparison. The minimum scenario lowered especially the carbon footprint of the WWT process. Maximum factors increased the result for WWT, energy and soil additives. When applying all minimum factors, the total footprint from both Viikinmäki and Metsäpirtti processes decreased with 33% and with maximum factors increased with 44%. High variation in the minimum and maximum values does not, however, address the distribution of the possible results.

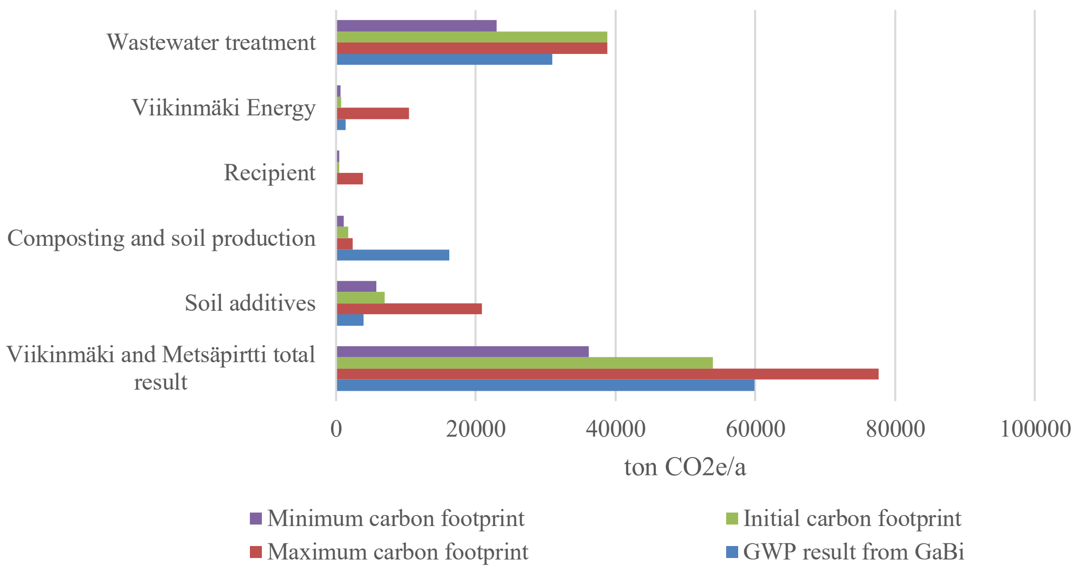

Figure 29: Variation of the results from the Carbon Footprint Calculation Tool applying different available emission factors, presented with GWP results from GaBi

9 Interpretation of Results

In this chapter, the results from the LCIA are discussed and recommendations are given. The main points are also summarized in ***Table 16***.

9.1 Biggest Contributors in Viikinmäki Result

The normalization of results to EU-level was used to highlight the most important environmental impact categories and thus the greatest factors increasing the overall environmental impact potential. The most relevant categories affected by the Viikinmäki process were EP, GWP and ADP (fossil). The three most important factors in the Viikinmäki process contributing to these impact categories were effluent nitrogen load, N_2O emissions and methanol production. It should be noted that all these three factors were linked to nitrogen removal.

Nitrogen load in effluent was especially important in EP category, which was the most relevant category in the normalized results. Phosphorous was only the third biggest contributor in the category after nitrogen and COD. In the used CML method, the phosphate equivalent factor was 0.42 for nitrogen and 3.06 for phosphorus which means the same amount of phosphorus produces a higher impact than nitrogen. As year 2018 was especially low for Viikinmäki in effluent phosphorus content, the contribution of phosphorus to the result could be higher calculated for other years. However, even calculated with year 2017 data, the contribution of phosphorus was still notably lower than that of nitrogen and COD load. Dominance of nitrogen load in EP was also noted in previous studies by Niero *et al.* (2014) and Tenhunen *et al.* (2000). The importance of the reduction of nitrogen in WWT in the Finnish context is also supported by Leppänen *et al.* (2012, p. 81), according to whom nitrogen is the limiting factor of growth in the Gulf of Finland, where the effluent of Viikinmäki WWTP is released. These factors underline the importance of further reduction of nitrogen in WWT.

In Viikinmäki, nitrogen is removed mainly in the activated sludge process and the reduction is enhanced with biological filtration. Optimizing the process parameters further for nitrogen removal could then decrease the environmental impact.

The treatment of reject water in scenario 5 could reduce nitrogen load from the WWT process and thus help optimize the process for nitrogen removal. However, estimation of the magnitude of this effect is speculative and thus it was not considered in the calculation. The treatment process of reject water itself did not cause big changes in the environmental impact of Viikinmäki process. In the scenario 5, the consumption of chemicals decreased causing a reduction in most impact categories. However, the two most important categories in normalization, EP and GWP slightly increased with a possible increase in N_2O emissions which was one of the three sub-scenarios modelled for reject treatment as the magnitude on N_2O emissions is uncertain. Overall, the total change in results was low. In the future, treatment of reject water is assumed to maintain the current level of nitrogen in effluent as influent nitrogen content increases. Treatment of reject water could hence help keep the environmental impact from rising in the future, making the scenario viable.

Table 16: Summarized results and recommendations of the study

Factor / Scenario	Impact in result	Possibilities for reduction of impact	To note
Effluent nitrogen	Positive, high	Optimizing N removal, possibly treating reject in scenario 5	
N_2O emissions from WWT	Positive, high	Optimizing N removal, possibly treating reject in scenario 5	
Methanol production for WWT	Positive, high	Change to alternative, e.g. bio-ethanol in scenario 4	Use assists in N removal, availability of alternative chemical must be checked, waste-based alternatives could be studied
Energy	Positive, low		Impact lowered by energy production on-site and purchasing renewable energy
Effluent phosphorus	Positive, low	Optimizing the process and adding treatment step from scenario 2	Effect on the total impact could be higher if studied for other years
Scenario 1	Positive, medium	Importing alternative chemical	If availability issues with the current chemical arise
Scenario 2	Altering, low		Could be beneficial as more P is removed, beneficial impact could be higher for other years with higher P load in effluent
Scenario 3, mainly PAC production	Positive, very high	Using alternative method or bio-based AC	Availability of alternative material must be checked
Scenario 4	Negative, high		Availability of chemical must be checked, also waste-based chemical could be studied
Scenario 5	Negative, low		Effect on N2O emissions unknown, could be beneficial for WWT process
Peat production for composting	Positive, high	Change to alternative material	No good material found yet
N2O emissions from composting	Positive, high	Optimizing the composting process	Ongoing project
Sand production for composting	Positive, medium	Change to recycled sand	

Improvement of nitrogen removal also motivates the use of an external carbon source in in the biological filtration. In Viikinmäki, this carbon source is methanol which, however, was the most important factor causing ADP (fossil) from the Viikinmäki process. According to normalization, EP from Viikinmäki had a higher share than ADP (fossil) in the EU-context but also the latter did not fall far behind. The use of methanol is thus bilateral. Changing methanol produced from natural gas to a bio-based option in scenario 4 decreased ADP (fossil) with roughly 50%. HTP was doubled in the scenario but was still relatively unimportant in the total normalized results. These outcomes indicated that a bio-based option for methanol could be viable considering environmental impact. The alternative chemical studied was produced from wheat, but it could also be produced from waste material. A waste-based carbon source could also be studied for even higher reduction of environmental impact potential. Important questions in considering change of chemical, however, are also reliable availability and cost.

Besides the nitrogen load in the effluent, the N_2O emissions from the WWT process had a big contribution to the total environmental impact of the Viikinmäki process as they were the biggest factor increasing GWP. N_2O was also the biggest contributor according to the Carbon Footprint Calculation Tool. Dominance of N_2O in GWP and carbon footprint of a WWTP was also noted by Daelman *et al*. (2013) and Rodriguez-Garcia *et al*. (2012). These results again highlight the importance of a functioning nitrogen removal as optimizing the process is important for minimizing both nitrogen load in the effluent and N_2O emissions.

Treatment of reject water could theoretically decrease the amount of N_2O emissions by decreasing the nitrogen load in the WWT process and this way enabling a more optimal activated sludge process performance with a lower ratio of N_2O produced. However, as this decrease in ratio is uncertain, it was not considered in the calculations.

Scenarios 1 and 3 had relevant impacts on the total results. According to the results from scenario 1, changing the precipitation chemical from byproduct-based ferrous sulphate to ferric sulphate would increase the environmental impacts in most categories. Changing the precipitation chemical was studied due to possible future availability issues. If ferrous sulphate from by-product was no longer available, the production via synthesis route would also cause a relevant increase in the environmental impact. As the environmental impact from transports was not high, these results indicate that if by-product ferrous sulphate was no longer available in Finland, it could be more viable to import it rather than produce it as longer transportation distances would likely cause only minor increase in the total LCA results.

Scenario 3 had a very high environmental impact mainly due to PAC production from fossil sources. Also, the production of oxygen and ozone increased the environmental impact of the scenario. The increase was so substantial that it can be questioned if the removal of micropollutants from the water in the model would compensate for this change. Thus, adding the removal of micropollutants to the process could cause more negative than positive effects in EU-level normalized results. Similar findings were indicated by Høibye *et al.* (2008) and Wenzel *et al.* (2008) who considered also the reduction of micropollutants in effluent. In the studies by Høibye *et al.* and Wenzel *et al.,* the environmental impact with micropollutant removal was higher than for the initial process, although they stated the used factors were debatable. Despite the results, the removal of micropollutants may become obligatory in the future with possible restrictions in legislation. This motivates studying alternatives, for example a bio- or waste-based activated carbon, for lower environmental impacts. Also, it could be investigated if ozonation alone would be sufficient for micropollutant removal.

After this LCA analysis, the planned micropollutant removal process for Viikinmäki has changed for technical reasons: PAC was changed to GAC which can partly be regenerated. Using GAC instead of PAC would also eliminate the need for the additional precipitation and disc filtration. These changes to the process would assumedly cause lower environmental impacts than the process utilizing PAC. Production of the GAC, however, would still cause high environmental impacts. Assessing the magnitude of the decrease in the results and weighting the importance of different categories in a local context is needed to further analyze positive and negative impacts of micropollutant removal with the planned process containing GAC.

9.2 Biggest Contributors of Metsäpirtti Result

According to the normalized results, the most relevant potential environmental impacts caused by the Metsäpirtti process were depletion of fossil resources and global warming. The use of peat was the main contributor to ADP (fossil) and the total impact for the Metsäpirtti process. The importance of peat was also visible in the carbon footprint results where it caused even a 156% bigger impact than in the GaBi GWP results. Peat is added in Metsäpirtti mainly as a supporting substance to the composting process but also some is used for improving the soil quality (Nipuli 2019). Decreasing the use of peat or completely changing it to some other material could lower the total environmental impact of composting and soil production notably. According to Nipuli (2019), HSY has already experimented with possible peat replacing products like woodchips, sawdust and fiber suspension. These materials, however, have not shown to work as well as peat: The compostability and quality of the compost and soil with these materials have been unsatisfactory. These results indicate a need for new studies on other possible materials considering the environmental impact of the process.

As with the Viikinmäki process, the N_2O emissions were one of the biggest factors contributing to the environmental impact of Metsäpirtti process by raising GWP and EP. They had, however, a minor impact in the carbon footprint calculation results due to the smaller amount of emissions assumed in the tool. Important sources of environmental impact from a similar type composting process were not discussed in exploited literature. Considering the results from the full LCA, minimization of GHG emissions from composting could decrease GWP and thus the total environmental impact of Metsäpirtti process substantially. Study on the effect of operational factors on GHG emissions from composting is ongoing inside HSY. With the results, N_2O emissions and the environmental impact of composting could be decreased.

The production of sand also caused and environmental impact, mainly in MAETP category. MAETP was the fourth biggest category in the normalized Metsäpirtti results. Sand is used for soil production. Use of some excess mineral material instead of virgin sand could be investigated to decrease MAETP from soil production.

9.3 Factors with Low Impacts

Energy production on-site kept the impact from energy-category moderate for the Viikinmäki process, although energy did cause the biggest rise to AP due to nitrogen and sulphur oxides emissions from on-site electricity production. This was different from the findings by Tenhunen *et al.* (2000) who stated that AP in their study was caused by effluent load.

Excess heat production caused a negative impact due to assumed avoidance of fossil fuel consumption. In Helsinki, heat is mostly produced as district heat at combined heat and power (CHP) plants which is more efficient than producing only heat. Thus, the negative environmental impact in the context of Helsinki is probably a bit lower than in the calculated results. Some of the heat, however, is also produced with fossil fuels at heat plants in situations when CHP plants do not offer enough heat. In year 2018, 91% of the heat consumed in Helsinki was produced as district heat, 9% of which was produced at heat-only plants (Helen Oy 2019).

The choice of eco-electricity for the purchased energy also kept the overall result low for both Viikinmäki and Metsäpirtti processes. Energy had an increasing effect mainly in POCP due to emissions from engines. These results were different from many previous studies where energy was either produced off-site or without completely renewable sources (Gustavsson & Tumlin 2013; Niero *et al*. 2014, Raghuvanshi *et al*. 2017). In these studies, energy was the biggest cause of environmental impacts. Even if the electricity purchased for Viikinmäki and Metsäpirtti were of fossil origin, the environmental impacts would have been a bit higher in the full LCA and only up to 1% higher in the carbon footprint calculation, depending on energy source.

Even though the environmental impact from electricity production was low, the Viikinmäki plant consumes a substantial amount of energy. Also, the amount of electricity produced from renewable sources does not cover all electricity consumption and thus there is not enough renewable energy for all consumers. According to the Official Statistics of Finland (2018), around 46% of electricity consumption in 2018 was covered with renewable sources. For these reasons, saving energy and producing more than is consumed at the plant is pursuable in a bigger context irrespective of the LCA results.

Other smaller impactors were for example other chemicals than methanol and transports which had a low impact in the total result. Solutions such as changing truck types to ones with lower emissions would be possible to execute but would not have a big impact on the overall results. From the studied scenarios, scenarios 2, effluent polishing, and 5, reject water treatment, did not cause big changes to the total results. Scenario 2 could, however, improve the treatment results. The results for scenario 2 could be even better if calculated for different years, as year 2018 was exceptionally good in terms of phosphorus removal. Thus, the difference in the treatment result with scenario 2 could be bigger for other years. Scenario 5 could also decrease the nitrogen load of the WWT process. Therefore, both scenarios 2 and 5 could be beneficial even though the LCA results indicated no change. Effluent polishing in scenario 2 may also become obligatory if treatment requirements become stricter.

The total environmental impact of the Metsäpirtti process would have been lower if the end use of sludge products was included in the full LCA study as it could reduce the need for fertilizer production. In the separate carbon footprint calculation, the end use was included but it had a very low effect on the results. However, it could have a more relevant effect in other impact categories. In previous research, the end use of sludge products was found to have a decreasing effect on EP (Niero *et al.* 2014) and an increasing effect on toxicity potentials (Corominas *et al.* 2013; Hong *et al.* 2008).

Also, including the end-of-life treatment of waste could slightly impact the full LCA total results of the Viikinmäki and Metsäpirtti processes. However, due to the fairly small amount of waste from the processes, the change would probably be low. This is supported by the carbon footprint calculation results where the waste treatment included was a very minor contributor to the total result.

9.4 Reliability of the Results

Relevance of the different impact categories were assessed with normalization of the results by comparing them to the total results of the EU. Normalization includes uncertainties but was found the most suitable method for handling the results in this study. Weighting of the results could offer more perspective on the most important results by comparing the importance of different impact categories and changes caused by alternative scenarios. In weighting, the results from different impact categories are related to weighting factors and summed into one total result. Choice of appropriate weighting factors, however, fell out of the scope of this study. Existing weighting factors are also not designed for the local context.

Both the full LCA and the carbon footprint study included uncertainty factors that could be studied further, but sensitivity analyses conducted for the categories and factors with the biggest normalized impacts indicated quite trustworthy results. The local parameter sensitivity analysis and Monte Carlo analysis showed low uncertainty when testing the five biggest contributing factors of the results. The uncertainty of the EP result was studied separately using another method. The results were of the same magnitude and thus quite reliable. The finding was in line with results from a study by Renou *et al.* (2008). The results from GaBi and the Carbon Footprint Calculation Tool were close to each other for the Viikinmäki process but quite different for the Metsäpirtti process. However, even with different results, nitrous oxide emissions and peat production were the biggest impactors from the Metsäpirtti process in both results.

9.5 Comparability and Generalization of the Total Results

Comparing the performance of Viikinmäki and Metsäpirtti processes with previous studies is limited due to different process types, system boundaries and LCA methods. However, comparing the magnitudes of the total numerical results allows for further estimation of their reliability and possibilities of generalization. The total numerical results of the Viikinmäki and Metsäpirtti processes in the three most relevant categories, EP, GWP and ADP (fossil) are presented in ***Table 17*** together with numerical, scaled total results from the reviewed LCA studies on WWT.

The overall numerical results of Viikinmäki and Metsäpirtti processes were generally higher than in the other studies reviewed (see ***Table 17***). The Viikinmäki and Metsäpirtti total result was only lower than GWP in the study by Raghuvanshi *et al.* (2017) and GWP and EP in the study by Aaltonen *et al.* (2014). In studies including also infrastructure, the result was both higher (Aaltonen *et al.* 2014) and lower (Buonocore *et al.* 2016) than the Viikinmäki and Metsäpirtti result. This indicates a variability in the results with different processes, although the exclusion or inclusion of infrastructure in the system boundaries might have a clearer effect in the results of other impact categories.

The performance of Viikinmäki and Metsäpirtti processes in terms of environmental impacts can be best compared with the results from the study by Gustavsson & Tumlin (2013) as they utilized the same system boundaries and the same Carbon Footprint Calculation Tool as this study. In the study by Gustavsson & Tumlin, carbon footprints were calculated for 16 WWTPs from the Nordic countries. Divided to impact by treated PE, the total result from Viikinmäki and Metsäpirtti processes was 50,7 kg CO_{2e}/PE/year, which was around 11% bigger than the 46 kg CO_{2e}/year average of the study. Part of this difference, however, can be explained with modification of the tool e.g. adding transports and soil additives. If peat was removed from the calculation, the carbon footprint of Viikinmäki would have been 44 kg CO_{2e}/PE/year which is roughly 4% lower than the study average. This indicates that the plant was performing quite well in terms of carbon footprint compared to the plants in the study. However, the performance in other environmental impact categories might differ from the carbon footprint results.

Generally, the total results of Viikinmäki and Metsäpirtti processes were of similar scale than in the studied literature which indicates reliability of the magnitude of the results. However, the differences in the results get notably higher when scaled to bigger units. With this comparison, it seems that generalization of the numerical results of an LCA study is limited for especially full WWT processes. However, the similarities found with the previous research on the largest factors contributing to the environmental impact indicate that general guidelines from the studies could be extrapolated to other WWT processes with caution. When generalizing the results to other processes, it should be noted that the contributing factors, such as nutrient load, chemicals and emissions, should be of same quality and quantity with the initial research.

The scenarios studied could be better generalized to other WWT processes as in the scenario, only the effect of changing or adding one individual factor in the process is studied. When generalizing the results from the scenarios, however, the process part should again be similar with the initial study for reliable estimates. For example, the micropollutant removal step should include PAC and reject water treatment should be executed with deammonification.

Also, the location of the WWTP should be considered when comparing the results to another WWT process. The CML method used in the LCIA of this study is designed in Europe and thus the factors used in the calculation might be different for a different context. Although, the EP result calculated with TRACI method, designed for North America, was quite similar to the EP result calculated with CML method for Europe. In addition to possible differences in impact category results, the relevance of the different categories might differ for different locations. The normalization in this study was conducted based on the total results of the EU-countries and thus the results could be different elsewhere.

In general, it seems that for large and complicated processes with many parameters varying from previous studies, a new LCA study should be conducted for reliable results. However, when evaluating the environmental impact of a small part of the process, guiding results could be sought after from existing literature.

Table 17: Viikinmäki and Metsäpirtti results compared with results from reviewed studies considering WWT process

Study by	GWP/ Carbon footprint (kg CO_2 eq.)	EP (kg PO_4 eq)	FDP/ADP fossil (MJ)	Main differences in the study scope
FU of 1000 m3 influent				
Viikinmäki and Metsäpirtti total result	650 / 583*	4.1	3.4×10^3	
Niero *et al.* (2014)	170	2.0*	2.3×10^3	Incineration and agricultural use of sludge
Buonocore *et al.* (2016)	400	2.4	2.9×10^3	Construction included, transports not included, different dewatering technology, gasification and landfilling of sludge
Raghuvanshi *et al.* (2017)	667	0.82**	-	Different WWT methods, network operation and agricultural use of sludge included
FU of 1 PE/year				
Viikinmäki and Metsäpirtti total result	54 / 51*	0.3	290	
Gustavsson & Tumlin (2013)	46	-	-	Only carbon footprint calculated, average result from different plants with partially different treatment processes
FU of 1 year				
Viikinmäki and Metsäpirtti total result	6.0×10^7 / 5.4×10^7*	3.8×10^5	3.2×10^8	
Aaltonen *et al.* (2014)	1.3×10^8	6.6×10^5	7.0×10^7	Infrastructure included, ultrafiltration as tertiary treatment

*Value from the Carbon Footprint Calculation Tool
** only marine eutrophication
***only freshwater eutrophication

10 Summary and Conclusions

In this study, an LCA was conducted for the WWT process at the Viikinmäki plant and the sludge handling process in the Metsäpirtti composting and soil production facility. Also, the effect of alternative chemicals and possible future scenarios on the Viikinmäki results were investigated. The objective of this thesis was to estimate the most relevant environmental impacts from the Viikinmäki and Metsäpirtti processes, the factors causing them and possible ways of mitigating the impacts. Also, the effects of the future scenarios to the total environmental impact as well as sensitivity of the results and possibilities for their generalization were studied.

The LCA was carried out according to a standardized framework, first defining the goal and scope of the study, followed by inventory analysis, life cycle impact assessment and finally an interpretation of the results. The impact assessment was conducted by modelling the Viikinmäki and Metsäpirtti processes with GaBi software and utilizing the CML calculation method. To identify the most relevant environmental impacts from the processes, the results were normalized by calculating the share of each category result out of the EU total impact. The categories with the highest shares were considered the most relevant when interpreting the results. Sensitivity analysis was conducted with local parameter sensitivity analysis, Monte Carlo analysis and by comparing the result of the most relevant impact category to a result calculated with another method. Besides conducting a full LCA, a carbon footprint was calculated separately with an Excel tool for comparison.

Five future scenarios were studied for different motives: Changing the precipitation chemical was studied due to possible future availability issues. Scenarios of adding effluent polishing with precipitation and filtration as well as removing micropollutants were included due to anticipated restrictions in future WWT requirements. Change of methanol produced from fossil fuels as a carbon source to a bio-based alternative was again studied to reduce the environmental impact. The fifth scenario considered was treating all reject water separately as it was assumed an option with a low environmental impact for lowering the nitrogen load of the WWT process and the operational costs of the plant.

According to the normalized results, the most relevant potential environmental impacts from the Viikinmäki and Metsäpirtti processes were eutrophication, global warming and depletion of fossil fuels. The factors in the processes contributing the most to these impacts were the nitrogen load in effluent, the nitrous oxide emissions from the WWT and composting process and the production of peat for the composting process and the production of methanol for the biological filtration. The production of excess heat at the Viikinmäki plant reduced the total environmental impact.

Optimizing the WWT operation of nitrogen removal could lead to a decrease in the environmental impact of the Viikinmäki process as it could reduce both effluent nitrogen load and nitrous oxide emissions. The change of methanol to a bio-based alternative could also decrease the environmental impact from the Viikinmäki process without disturbing nitrogen removal. The total environmental impact from the Metsäpirtti process could be substantially reduced by optimizing the composting process for less nitrous oxide emissions and by changing peat to another appropriate material. Achieving lower N_2O emissions requires more knowledge on the factors influencing GHG emissions from the composting process which is currently an ongoing study at HSY. The total impact from the Metsäpirtti process could also be moderately decreased by replacing the use of virgin sand with a recycled alternative.

Adding micropollutant removal to the WWT process in the future scenarios increased environmental impact substantially due to activated carbon production from fossil origin. The change was so substantial that it can be questioned if the decreasing impact from micropollutant removal would be enough to compensate the impact from PAC. The WWT plant must, however, remove micropollutants if required by law in the future. For lower environmental impacts from micropollutant removal, waste-based options for activated carbon could be investigated. Besides micropollutant removal, also change of precipitation chemical caused a relevant increase in environmental impact in the future scenarios. Effluent polishing with precipitation and disc filtration as well as treatment of reject water caused little changes to the total Viikinmäki results. Their execution could be viable considering environmental impact of Viikinmäki process as they could improve the quality of effluent.

Sensitivity analyses conducted indicated low uncertainty of the results. However, results from this study should be treated as estimations due to the speculative nature of an LCA. When comparing the results of this study to findings from reviewed literature, it can be noted that the most important factors contributing to the total impact were often similar, but the total numerical results varied with different processes and system boundaries. Thus, the main findings from this study might be viable for generalization for WWT processes with similar process parameters and context. As WWT processes are always unique, the calculation results from individual process factors and the findings from the scenarios studied might be more easily extrapolated to other WWT processes than the total results. However, for accurate total numerical results, the study should be reproduced for the process in question.

References

Aaltonen, M. & Jantunen, A. & Timonen, J. & Bruce, T. 2014. Lappeenrannan jäte- ja jätevesiratkaisujen elinkaariarviointi. Wirma Lappeenranta oy. 62 p. Available at: https://www.wirma.fi/sites/default/files/Raportit_ja_dokumentit/lappeenrannan_jate-_ja_jatevesiratkaisujen_elinkaariarviointi_final_repo.pdf

Acero, A.P. & Rodriguez, C. & Ciroth, A. 2016. LCIA methods – Impact assessment methods in life cycle assessment and their impact categories. 1.5.5. Greendelta. Available at: http://www.openlca.org/wp-content/uploads/2016/08/LCIA-METHODS-v.1.5.5.pdf.

Althaus H.J. & Chudacoff M. & Hischier R. & Jungbluth N. & Osses M. & Primas A. 2007. Life Cycle Inventories of Chemicals – Data v2.0. Swiss Centre for Life Cycle Inventories. 876 p. Ecoinvent report No. 8. Available at: https://db.ecoinvent.org/reports/08_Chemicals.pdf

Antikainen, R. 2010. Elinkaarimetodiikkojen nykytila, hyvät käytännöt ja kehitystarpeet. Suomen ympäristökeskus. 83 p. Suomen ympäristökeskuksen raportteja 7/2010. ISBN 978-952-11-3734-1.

Boldrin, A. & Hartling, K.R. & Laugen, M. & Christensen, T.H. 2010. Environmental inventory modelling of the use of compost and peat in growth media preparation. Resources, conservation and recycling. Vol 54. ISSN 0921-3449. DOI 10.1016/j.resconrec.2010.04.003.

Bonton, A. & Bouchard, C. & Barbeau, B. & Jedrzejak, S. 2011. Comparative life cycle assessment of water treatment plants. Desalination. Vol 284. P. 42–54. ISSN 0011-9164. DOI 10.1016/j.desal.2011.08.035.

Brown, S. & Kruger, C. & Subler, S. 2008. Greenhouse gas balance for composting operations. Journal of environmental quality. Vol 37:4. P. 1396–1410. ISSN 0047-2425. DOI 10.2134/jeq2007.0453.

Buonocore, E. & Mellino, S. & De Angelis, G. & Liu, G. & Ulgiati, S. 2016. Life cycle assessment indicators of urban wastewater and sewage sludge treatment. Ecological indicators. Vol 94:3. P. 13–23. ISSN 1470-160X. DOI 10.1016/j.ecolind.2016.04.047.

Cao, Y. & Pawlowski, A. 2012. Life cycle assessment of two emerging sewage sludge-to-energy systems: Evaluating energy and greenhouse gas emissions implications. Bioresource technology. Vol 127. P. 81–91. ISSN 0960-8524. DOI 10.1016/j.biortech.2012.09.135.

Chae, K.J & Kang, J. 2013. Estimating the energy independence of a municipal wastewater treatment plant incorporating green energy resources. Energy conversion and management. Vol 75. P. 664–672. ISSN 0196-8904. DOI 10.1016/j.enconman.2013.08.028.

Chen, Y.C. 2019. Estimation of greenhouse gas emissions from a wastewater treatment plant using membrane bioreactor technology. Water environment research. Vol 91:2. P. 111–118. ISSN 1554-7531. DOI 10.1002/wer.1004.

Christensson, M. & Lie, E. & Welander, T. 1994. A comparison between ethanol and methanol as carbon sources for denitrification. Water science and technology. Vol 30:6. P. 83–90. ISSN 0273-1223.

Corominas, L. & Foley, J. & Guest, J.S. & Hospido, A. & Larsen, H.F. & Morera, S. & Shaw, A. 2013. Life cycle assessment applied to wastewater treatment: State of the art. Water research. Vol. 47:15. P. 5480–5492. ISSN 0043-1354. DOI 10.1016/j.watres.2013.06.049.

Daelman, M. & van Voorthuizen, E. & van Dongen, L. & Volcke, E. & van Loosdrecht, M. 2013. Methane and nitrous oxide emissions from municipal wastewater treatment – results from a long-term study. Water science and technology. Vol 67:10. P. 2350–2355. ISSN 0273-1223. DOI 10.2166/wst.2013.109.

European Commission (EC). European Platform on Life Cycle Assessment (LCA). [Cited 2.6.2019]. Available at: http://ec.europa.eu/environment/ipp/lca.htm

European Commission, Joint research centre (EC-JRC). 2019. ILCD International Life Cycle Data system. [Cited 2.6.2019]. Available at: https://eplca.jrc.ec.europa.eu/ilcd.html

European Commission, Joint research centre, Institute for environment and sustainability EC-JRC-IES. 2010a. International reference life cycle data system (ILCD) hanbook – Analysis of existing environmental impact assessment methodologies for use in life cycle assessment. Luxembourg: Publications office of the European Union. 72 p. Available at: https://eplca.jrc.ec.europa.eu/uploads/ILCD-Handbook-LCIA-Background-analysis-online-12March2010.pdf

European Commission, Joint research centre, Institute for environment and sustainability EC-JRC-IES. 2010b. International reference life cycle data system (ILCD) handbook – General guide for life cycle assessment. Luxembourg: Publications office of the European Union. 375 p. ISBN 978-92-79-19092-6. DOI 10.2788/38479.

European Commission, Joint research centre, Institute for environment and sustainability EC-JRC-IES. 2010c. International reference life cycle data system (ILCD) handbook – Specific guide for life cycle inventory data sets. Luxembourg: Publications office of the European Union. 129 p. ISBN 978-92-79-19093-3. DOI 10.2788/39726.

European Commission, Joint research centre, Institute for environment and sustainability EC-JRC-IES. 2011. International reference life cycle data system (ILCD) handbook – Recommendations for life cycle impact assessment in the European context – Based on existing environmental impact assessment models and factors. Luxembourg: Publications office of the European Union. 129 p. ISBN 978-92-79-17451-3. DOI 10.278/33030.

Frischknecht, R. & Althaus, H.J. & Bauer, C. Doka, G. & Heck, T. & Jungbluth, N & Kellenberger, D. & Nemecek, T. 2007. The environmental relevance of capital goods in life cycle assessments of products and services. The international journal of life cycle assessment. Vol 12. P. 7–17. ISSN 1614-7502. DOI 10.1065/lca2007.02.309.

Frischknecht, R. & Steiner, R. & Jungbluth, N. 2009. The ecological scarcity method – Eco-factors 2006 – A method for impact assessment in LCA. Bern, Switzerland: Federal Office for the Environment. 188 p. Environmental studies no. 0906. Available at: https://www.bafu.admin.ch/dam/bafu/en/dokumente/wirtschaft-konsum/uw-umwelt-wissen/methode_der_oekologischenknappheitoekofaktoren2006.pdf.download.pdf/the_ecological_scarcitymethodeco-factors2006.pdf.

Guereca, L.P. & Jato-Espino, D. & Lizasoain-Artega, E. 2019. Life cycle assessment of construction materials: analysis of environmental impacts and recommendations of eco-efficient management practices, in Hussain, C.M. (ed.) Handbook of environmental materials management. Springer international publishing. ISBN 978-3-319-73644-0. DOI 10.1016/j.buildenv.2010.12.002.

Guinée, J.B. (Ed.) & Gorrée, M. & Heijungs, R. & Huppes, G. & Kleijn, R. & de Koning, A. & van Oers, L. & Wegener Sleeswijk, A. & Suh S. & Udo de Haes, H.A. & de Bruijn, J.A. & van Duin, R. & Huijbregts, M.A.J. 2002. Handbook on life cycle assessment: Operational guide to the iso standards. Dordrecht, the Netherlands: Kluwer academic publishers. 692 p. Eco-efficiency in industry and science. ISBN 1-4020-0228-9.

Gustavsson, D.J. & Tumlin, S. 2013. Carbon footprints of scandinavian wastewater treatment plants. Water science and technology. Vol 68:4. P. 887–893. ISSN 0273-1223. DOI 10.2166/wst.2013.318.

Hauschild, M. (ed.) & Huijbregts, M. (ed.). 2015. Life cycle impact assessment. Dordrecht, the Netherlands: Springer. 330 p. LCA compendium: the complete world of life cycle assessment. ISSN 2214-3505. DOI 10.1007/978-94-017-9744-3_1.

Helen Oy. Liiketoimintavuosi. [Cited 26.11.2019]. Available at: https://www.helen.fi/vuosikertomus/vuosikertomus-2016/konsernin-vuosi/liiketoimintavuosi.

Hong, J. & Hong, J. & Otaki, M. & Jolliet, O. 2008. Environmental and economic life cycle assessment for sewage sludge treatment process in Japan. Waste management. Vol: 29:2. P. 696–703. ISSN 0956-053X. DOI 10.1016/j.wasman.2008.03.026.

Hospido, A. & Carballa, M. & Moreira, M. & Omil, F. & Lema, J.M. & Feijoo, G. 2010. Environmental assessment of anaerobically digested sludge reuse in agriculture: Potential impacts of emerging micropollutants. Water research. Vol 44:10. P. 3225–3233. ISSN 0043-1354. DOI 10.1016/j.watres.2010.03.004.

Hospido, A. & Sanchez, I. & Rodriguez-Garcia, G. & Iglesias, A. & Buntner, D. & Reif, R. & Moreira, M.T. & Feijoo, G. 2012. Are all membrane reactors equal from an environmental point of view? Desalination. Vol 285. P. 263–270. ISSN 0011-9164. DOI 10.1016/j.desal.2011.10.011.

Høibye, L. & Clauson-Kaas J. & Wenzel, H. & Larsen, H.F. & Jacobsen, B.N. & O. Dalgaard. 2008. Sustainability assessment of advanced wastewater treatment technologies. Water science and technology. Vol 58:5. P. 963–968. ISSN 0273-1223. DOI 10.2166/wst.2008.450.

Intergovernmental panel on climate change IPCC, ed. Eggleston, S. & Buendia, L & Miwa, K. & Ngara, T & Tanabe, K. 2006. Waste. Hayama, Japan: Institute for Global Environmental Strategies IGES. IPCC guidelines for national greenhouse gas inventories. ISBN 4-88788-032-4.

Langer, M. & Schermann, A. 2013. Feasibility of the microsieve technology for advanced phosphorus removal. Berlin, Germany: KompetenzZentrum Wasser Berlin gGmbH. 78 p.

Leppänen, J.M & Rantajärvi, E. & Bruun, J.E. & Salojärvi, J. (Ed.). 2012. Suomen merenhoitosuunnitelman valmisteluun kuuluva meriympäristön nykytilan arvio. 102 p. Available at: https://www.ymparisto.fi/download/noname/%7B62E78B95-3FCA-41ED-AC1C-DA865DF646AE%7D/34442.

Li, L. & Ling, Y. & Wang, H. & Chu, Z. & Yan, G. & Li, Z. &Tong, W. 2019. N_2O emission in partial nitritation-anammox process. Chinese chemical letters. ISSN 1001-8417. DOI 10.1016/j.cclet.2019.06.035.

Lundin, M. & Bengtsson, M. & Molander, B. 2000. Life cycle assessment of wastewater systems: Influence of system boundaries and scale on calculated environmental loads. Environmental science & technology. Vol 34:1. P. 180–186. ISSN 0013-936X. DOI 10.1021/es990003f.

Mattila, T. & Helin, T. & Antikainen, R. & Soimakallio, S. & Pingoud, K. & Wessman, H. 2011. Land use in life cycle assessment. Suomen ympäristökeskus. 86 p. Suomen ympäristö 24/2011. ISBN 978-952-11-3926-0.

Niero, M & Pizzol, M. & Bruun, H.G. & Thomsen, M. 2014. Comparative life cycle assessment of wastewater treatment in Denmark including sensitivity and uncertainty analysis. Journal of cleaner production. Vol. 68. P. 25–35. ISSN 0959-6526. DOI 10.1016/j.jclepro.2013.12.051.

Official Statistics of Finland (OSF). 2018. Production of electricity and heat. Helsinki: Statistics Finland. ISSN 1798-5099.

Open LCA. LCA data. [Cited 28.6.2019]. Available at: http://www.openlca.org/lca-data/.

Pagans, E. & Barrena, R. & Font, X. & Sánchez. 2005. Ammonia emissions from the composting of different organic wastes. Dependency on process temperature. Chemosphere. Vol 62. P. 1534–1542. ISSN 0045-6535. DOI 10.1016/j.chemosphere.2005.06.044.

Pitter, P. & Chudoba, P. 1990. Biodegradability of organic substance in the aquatic environment. Boston, Illinois, USA: CRC Press. 306 p. ISBN 0-8493-5131-6.

Pizzol, M. & Christensen, P. & Schmidt, J. & Thomsen, M. 2011. Impacts of "metals" on human health: a comparison between nine different methodologies for life cycle impact assessment (LCIA). Journal of cleaner production. Vol 19:6–7. P. 646–656. ISSN 0959-6526. DOI 10.1016/j.jclepro.2010.05.007.

Raghuvanshi, S. & Bhakar, V. & Sowmya, C. & Sangwan, K.S. 2017. Waste water treatment plant life cycle assessment: treatment process to reuse of water. Procedia CIRP. Vol 61. P. 761–766. ISSN 2212-8271. DOI 10.1016/j.procir.2016.11.170.

Rodriguez-Garcia, G. & Hospido, A. & Bagley, D.M. & Moreira, M.T. & Feijoo, G. 2012. A methodology to estimate greenhouse gases emissions in life cycle inventories of wastewater treatment plants. Environmental impact assessment review. Vol 37. P. 37–46. ISSN 0195-9255. DOI 10.1016/j.eiar.2012.06.010.

Sánchez-García, M. & Roig, A. & Sánchez-Monedero, M.A. & Cayuela, M.L. 2014. Frontiers in environmental science. Vol 25. ISSN 2296-665X. DOI 10.3389/fenvs.2014.00025.

SFS-EN ISO 14040. 2006. Ympäristöasioiden hallinta. Elinkaariarviointi. Periaatteet ja pääpiirteet. 2nd edition. Finnish standards association. 48 p.

SFS-EN ISO 14044. 2006. Ympäristöasioiden hallinta. Elinkaariarviointi. Vaatimukset ja suuntaviivoja. 2nd edition. Finnish standards association. 109 p.

SimaPro. LCI databases in SimaPro. [Cited 28.6.2019]. Available at: https://simapro.com/databases/.

Solomon, S. & Daniel, J.S. & Sanford, T.J. & Murphy, D.M. & Plattner, G.K. & Knutti, R. & Friedlingstein, P. 2010. Persistence of climate changes due to a range of greenhouse gases. Proceedings of the national academy of sciences of the United States of America – PNAS. Vol 107:43. P. 18354–18359. ISSN 1091-6490. DOI 10.1073/pnas.1006282107.

Tenhunen, J. & Oinonen, J. & Seppälä, J. 2000. Vesihuollon elinkaaritutkimus – Tampereen vesilaitoksen vaikutukset ympäristöön. Suomen ympäristökeskus. 107 p. Suomen ympäristö 434. ISBN 952-11-0784-7.

Thinkstep. GaBi software – to drive product sustainability! [Cited 10.7.2019a]. Available at: http://www.gabi-software.com/international/software/gabi-software/.

Thinkstep. LCA databases. [Cited 28.6.2019b]. Available at: https://www.gabi-software.com/nw-eu-english/databases/.

Thinkstep. Q & A webinar April 24. [Cited 15.10.2019c] [video]. Available at: http://www.gabi-software.com/support/gabi-learning-center/questions-answers/.

Time for change. CO2 emissions for shipping of goods. [Cited 10.8.2019] Available at: https://timeforchange.org/co2-emissions-for-shipping-of-goods/

Tseng, L. & Robinson, A. & Zhang, X. &Xiaomei, X. & Southon, J. & Hamilton, A. & Sobhani, R. & Stenstrom, M. & Rosso, D. 2016. Identification of preferential paths of fossil carbon within water resource recovery facilities via radiocarbon analysis. Environmental science and technology. Vol 50:22. P. 12166–12178. ISSN 1520-5851. DOI 10.1021/acs.est.6b02731.

VA-teknik Södra. Klimatpåverkan – beräkningsverktyg. [Cited 11.7.2019]. Available at: https://va-tekniksodra.se/klimatpaverkan-berakningsverktyg/.

Van Oers, L. & de Koning, A. & Guinée, J.B. & Huppes, G. 2002. Abiotic resource depletion in LCA. Road and hydraulic engineering institute. Available at: https://www.leidenuniv.nl/cml/ssp/projects/lca2/report_abiotic_depletion_web.pdf

Visser, C. & Odegard, I.Y.R. & Naber, N.R. & Bergsma, G.C. & Nieuwenhujizen, A.F. & Sanders, M.H.A. 2016. Levenscyclusanalyse van grondstoffen uit rioolwater. Amersfoort, Netherlands: Stichting Toegepast Onderzoek Waterbeheer. 100 p. STOWA 2016-22. ISBN 978.90.5773.713.8.

Wenzel, H. & Larsen, H.F. & Clauson-Kaas, J. & Høibye, L. & Jacobsen B.N. 2008. Weighing environmental advantages and disadvantages of advanced wastewater treatment of micro-pollutants using environmental life cycle assessment. Water science and technology. Vol 57:1. P. 27–32. ISSN 0273-1223. DOI 10.2166/wst.2008.819.

World Nuclear Association. Heat values opf various fuels. [Cited 13.12.2019] Available at: https://www.world-nuclear.org/information-library/facts-and-figures/heat-values-of-various-fuels.aspx

Ymparisto.fi. Yhdyskuntien jätevesien kuormitus vesiin. [Cited 13.12.2019] Available at: https://www.ymparisto.fi/fi-FI/Kartat_ja_tilastot/Vesihuoltoraportit/Yhdyskuntien_jatevesien_kuormitus_vesiin

Zhang, J. & Yuan, H. & Deng, Y. & Zha, Y. & Abu-Reesh, I.M. & He, Z. & Yuan, C. 2018. Life cycle assessment of a microbial desalination cell for sustainable wastewater treatment and saline water desalination. Journal of cleaner production. Vol 200. P. 900-910. ISSN 1879-1786. DOI 10.1016/j.jclepro.2018.07.197.

Miscellaneous Sources

Aurola, A.M. 2019. Nordkalk Corporation. Personal communication. 23.8.2019.

Juntunen, A. 2019. Yara. Personal communication. 21.8.2019.

Kettunen, V. 2019. Kemira. Personal communication. 23.9.219.

Lindholm, J. 2019a. NovoChem. Personal communication. 20.8.2019.

Lindholm, J. 2019b. NovoChem. Personal communication. 23.8.2019.

Nipuli, J. 2019. Helsinki Region Environmental Services Authority HSY. Personal communication. 18.7.2019.

List of Appendices

Appendix 1: LCI for the Viikinmäki Process

Input	Amount	Unit	Changed for scenario	Amount in scenario	Unit
External sludge	66740000	kg			
Ferrous sulphate from byproduct	8902700	kg	1a and 1b	0	kg
Ferrous sulphate via synthesis	0	kg	1a	8902700	kg
Ferric sulphate	0	kg	1b	14163386	kg
			2 and 3	97090	kg
Calcium Hydroxide	2040600	kg	5	1493100	kg
Tap water	19071	m3			
Methanol	2532700	kg	4	0	kg
			5	2226100	kg
Bio-ethanol	0	kg	4	1817727	kg
Electricity purchased	1113	MWh	2	1686	MWh
			3	11645	MWh
Oxygen	0	kg	3	7488093	kg
PAC	0	kg	3	1456355	kg
Polymer	125468	kg	2 and 3	1776004	kg

Output	Amount	Unit	Changed for scenario	Amount in scenario	Unit
Screenings	555000	kg			
Sand to landfill	300000	kg			
Sandy screenings	16000	kg			
Waste from real-estate	102000	kg			
COD to sea	4006970	kg			
Suspended solids to sea	19071	kg			
Nitrogen to sea	454060	kg			
Phosphorus to sea	15002	kg	2 and 3	9709	kg
1,1,1-Trichloroethane to air	1	kg			
Ammonia to air	1998	kg			
Benzene to air	11	kg			
Carbon dioxide (biotic) to air	33463778	kg			
Dichloroethane to air	1	kg			
Dicloromethane to air	1	kg			
Methane to air	218010	kg			
Nitrogen oxides to air	1159	kg			
Nitrous oxide to air	93645	kg	5b	107692	kg
			5c	86622	kg
NMVOC to air	2985	kg			
Sulphur oxides to air	5	kg			

Appendix 2: LCI for the Composting and Soil Production Process in Metsäpirtti

Input	Amount	Unit
Dried sludge from Viikinmäki	60325	t
Dried sludge from Suomenoja	12879	t
Peat	17450	t
Biotite	498	t
Sand	39450	t
Horse manure	5120	t
Coffee waste	218	t
Tap water	1025	t
Light fuel oil	114913	l
Electricity	144	MWh

Output	Amount	Unit
Compost	115216000	kg
Waste	22180	kg
NH_3 to air	14641	kg
CO_2 (biotic) to air	14787208	kg
CH_4 to air	95165	kg
N_2O to air	51243	kg

Appendix 3: LCIs for the Production Processes of Chemicals

Chemical	Flow direction	Flow	Amount	Unit	Source
Ferric sulphate	Inputs	Ferrous sulphate	653	kg	Kettunen 2019
		Oxygen	23	kg	
		Sulphuric acid 96%	78	kg	
		Water	250	kg	
		Electricity	16	kWh	
		Steam	0.022	MWh	
	Outputs	Ferric sulphate	1000	kg	
Methanol	Inputs	Natural gas	32.7	MJ (net)	Althaus *et al.* 2007, table 53.11
		Process and cooling water	9.01	kg	
		Electricity	0.27	MJ	
	Outputs	Methanol	1	kg	
		Waste heat	13.9	MJ	
		Cooling water	6.36	kg	
		CO_2 to air	0.431	kg	
		NO_x to air	0.00033	kg	
		SO_x to air	0.000018	kg	
		CH_4 to air	0.00098	kg	
		CH_3OH to air	0.00053	kg	
		COD to fresh water	0.00049	kg	
		DOC to fresh water	0.00024		
		AOX to fresh water	0.000001	kg	
		P to fresh water	0.00001	kg	
		CH_2O to fresh water	0.0001	kg	
		CH_3OH to fresh water	0.00003	kg	
		C_6H_6 to fresh water	0.00001	kg	
		SS to fresh water	0.00002	kg	
		Chloride to fresh water	0.000002	kg	
PAC	Inputs	Deionized water	12	kg	Zhang *et al.* 2018, table S8
		Heat from hard coal	60.8	MJ	
		Heat from natural gas	13.2	MJ	
		Electricity	9224	MWh	
		Hard coal	1	kg	
	Outputs	PAC	1	kg	
		Waste heat	79.8	MJ	
Ozone	Inputs	Oxygen	7.7	g	HSY
		Electricity	9.5	Wh	
	Outputs	Ozone	1	g	

Appendix 4: LCIs for the Production Processes of Compost and Soil Additives

Additive	Flow direction	Flow	Amount	Unit	Source
Biotite	Inputs	Electricity	0,002	MWh	Juntunen 2019/ estimated
		Light fuel oil	8	l	
	Outputs	Biotite	1000	kg	
		C_2O	21.339	kg	
		CH_4	0.0009	kg	
		N_2O	0.0002	kg	
Peat	Inputs	Peat	1000	kg	Boldrin *et al.* 2010, table 7
		Diesel	5.7	kg	
		LPG	2.1	kg	
		Gasoline	0.73	kg	
		Light fuel oil	0.02	kg	
		Natural gas	$3.69*10^{-6}$	kg	
	Outputs	Peat	1000	kg	
		CO_2 (fossil) to air	13.9	kg	
		NO_x to air	0.48	kg	
		CH_4 to air	198.8	g	
		N_2O to air	14	g	
		CO to air	171	g	
		HC to air	70	g	
		Dust to air	50	g	
		SO_2 to air	38	g	
		Organic matter to water	3	kg	
		COD_{Mn} to water	1.8	kg	
		SS to water	1.7	kg	
		N to water	180	g	
		NH_4-N to water	86	g	
		P to water	5.8	g	

Appendix 5: Emission Factors Used in Carbon Footprint Calculation for Different Transport Vehicles

Vehicle and load	Emission factor (g CO_{2e} /km)	Source
Truck		HSY
Empty	458	
50% load	508	
Full load	559	
Semi-trailer		HSY
Empty	766	
70% load	959	
Full load	1041	
Trailer		HSY
Empty	831	
70% load	1132	
Full load	1260	
Ferry	51	HSY
Train	30	(Time for change 2019)

Appendix 6: LCIs and Vehicle Types Used for Different Transport Processes

Transported material	Process in GaBi	Type in carbon footprint calculation	Distance (km)	Load (t), 50% for road transport in GaBi
Viikinmäki				
External sludge	Truck, euro 0 - 6 mix, 7.5 t - 12t gross weight	Truck 50% load + empty	30	4.7
Sand waste	Truck, euro 0-6 mix, 12-14 gross weight	Truck full load + empty	30	9.0
Screenings	Truck, euro 0-6 mix, 12-14 gross weight	Truck full load + empty	13	8.3
Sand screenings	Truck, euro 0-6 mix, 12-14 gross weight	Truck 50% load + empty	30	3.6
Methanol (train)	Rail transport cargo-Diesel, extra large train, gross tonne weight 2,000t	Modern train	4 568	2 226.1
Methanol (road)	Truck-trailer, Euro 0-6 mix, 50-60t gross weight	Trailer 70% load + empty	142	30.5
Ferrous sulphate	Truck-trailer, Euro 6, 34-40t gross weight	Trailer full load + empty	254	37.7
Calcium hydroxide	Truck-trailer, Euro 0 - 6 mix, 34 - 40t gross weight	Trailer full load + empty	95	45.3
Polymer (ferry)	Bulk commodity carrier, average, ocean going	Ferry	2 500	125.5
Polymer (road)	Truck-trailer, Euro 0 - 6 mix, 28 - 34t gross weight	Semi-trailer 70% load + empty	130	15.0
Light fuel oil	Truck-trailer, Euro 0 - 6 mix, 34 - 40t gross weight	Semi-trailer 70% load + empty	50	16.2

Continues on the next page

Continues from the previous page

Transported material	Process in GaBi	Type in carbon footprint calculation	Distance (km)	Load (t), 50% for road transport in GaBi
Viikinmäki				
Sludge to Metäpirtti	Truck-trailer, Euro 0 - 6 mix, 34 - 40t gross weight / 27t payload capacity	Trailer full load + empty	35	55.0
Sludge to other facility	Truck-trailer, Euro 0 - 6 mix, 34 - 40t gross weight	Trailer full load + empty	43	55.0
Metsäpirtti				
Horse manure	Truck-trailer, Euro 0 - 6 mix, up to 28t gross weight	Semi-trailer 70% load + empty	35	9.9
Sand	Truck-trailer, Euro 6, 50 - 60t gross weight	Trailer full load + empty	71	53.0
Biotite	Truck-trailer, Euro 6, 50 - 60t gross weight	Trailer full load + empty	408	41.5
Peat	Truck-trailer, Euro 6, 34 - 40t gross weight	Trailer full load + empty	169	32.0
Coffee waste	Truck, euro 0-6 mix, 12-14 gross weight	Truck full load + empty	28	7.8
Products delivered	Truck-trailer, Euro 0 - 6 mix, 34 - 40t gross weight	Trailer 70% load + empty	43	27.6
Products collected	Truck, Euro 0 - 6 mix, 28 - 32t gross weight	Truck 50% load + empty	35	17.3
Waste	Truck, euro 0-6 mix, 12-14 gross weight	Truck full load + empty	24	7.8
Sludge from Suomenoja WWTP	Truck-trailer, Euro 0 - 6 mix, 34 - 40t gross weight	Trailer full load + empty	58	53.9

Appendix 7: LCI for the On-Site Energy Production

Input	Amount	Unit
Light fuel oil	19	m^3

Output	Amount	Unit
Heat	11126	MWh
Methane to air	80712	kg
Carbon monoxide to air	82738	kg
Nitrogen oxides to air	33376	kg
Sulphur oxides to air	12994	kg
Dust to air	59	kg
Carbon dioxide (biogenic) to air	26942217	kg
Carbon dioxide (fossil) to air	49620	kg

Appendix 8: Numeric LCIA results for the Viikinmäki process divided for different factors

Category	ADP (fossil) (MJ)	AP (kg SO_2 eq.)	EP (kg PO_4 eq.)	FAETP (kg DCB eq.)	GWP* (kg CO_2 eq.)	HTP (kg DCB eq.)	MAETP (kg DCB eq.)	POCP (kg C_2H_4 eq.)	TETP (kg DCB eq.)
WWT	81615	3794	26137	182	30928095	24154	681631	1799	26
Nitrogen oxides emissions	0	580	25451	0	24815925	1391	0	33	0
Methane emissions	0	0	0	0	6104280	0	0	1308	17
Other organic emissions to air	0	0	0	148	1973	22027	46	457	0
Other inorganic emissions to air	0	3200	699	0	0	200	0	0	0
Tap water	81615	12	3	34	5917	536	681593	1	9
Chemicals	114274770	5535	705	77979	4638118	26996	88466955	846	5112
Calcium hydroxide	5346794	441	88	541	1873205	13361	18329505	19	4990
Ferrous sulphate	217551	91	23	88	16878	561	220312	8	28
Methanol	98448012	3330	411	77292	2337407	11591	61172133	560	87
Polymer	10262413	1673	184	59	410629	1483	8745005	260	8

Continues on the next page

Continues from the previous page

Category	ADP (fossil) (MJ)	AP (kg SO_2 eq.)	EP (kg PO_4 eq.)	FAETP (kg DCB eq.)	GWP* (kg CO_2 eq.)	HTP (kg DCB eq.)	MAETP (kg DCB eq.)	POCP (kg C_2H_4 eq.)	TETP (kg DCB eq.)
Energy	-8457941	31442	4467	1755	1410921	17942	-39273974	2054	-287
Boilers	816181	130	27	266	57250	1299	761268	13	7
Engines	0	32161	4309	0	2208808	41050	0	2025	0
Heat production	-9729458	-2529	-222	-529	-981325	-41537	-63335288	-156	-570
External electricity production	455336	1681	353	2018	75088	17121	23300045	95	276
Torch	0	0	0	0	51100	10	0	77	0
Transports	17953765	2679	652	7480	1333064	36004	19409707	-745	2291
Calcium hydroxide	185190	21	5	77	13823	362	200225	-4	24
External sludge	12445768	1970	483	5186	135646	25169	13456184	-589	1589
Ferrous sulphate	2566323	383	94	1069	191255	5058	2774671	-104	328
Light fuel oil	1301	0	0	1	97	3	1407	0	0
Methanol	802226	52	12	334	59677	1561	867355	0	102
Waste	77930	11	3	33	15806	159	84257	-4	10
Polymer	54410	36	4	21	4146	134	57183	0	5
Sludge to composting	1820617	204	50	758	171201160 55	3558	1968425	-44	232

Continues on the next page

Continues from the previous page

Category	ADP (fossil) (MJ)	AP (kg SO_2 eq.)	EP (kg PO_4 eq.)	FAETP (kg DCB eq.)	GWP* (kg CO_2 eq.)	HTP (kg DCB eq.)	MAETP (kg DCB eq.)	POCP (kg C_2H_4 eq.)	TETP (kg DCB eq.)
Sludge treatment	4279	1	0	2	114032	28	35735	24	0
Real-estate	12164	2	0	5	882	80	101587	0	1
Recipient	0	0	324685	0	0	0	0	0	0
Nitrogen	0	0	190659	0	0	0	0	0	0
Phosphorus	0	0	45895	0	0	0	0	0	0
COD	0	0	88132	0	0	0	0	0	0

Appendix 9: Numeric LCIA results for the Metsäpirtti process divided for different factors

Category	ADP (fossil) (MJ)	AP (kg SO_2 eq.)	EP (kg PO_4 eq.)	FAETP (kg DCB eq.)	GWP* (kg CO_2 eq.)	HTP (kg DCB eq.)	MAETP (kg DCB eq.)	POCP (kg C_2H_4 eq.)	TETP (kg DCB eq.)
Composting and soil production	5272	23426	18960	2	16244397	1499	44024	571	1
Nitrous oxide	0	0	13 800	0	13579401		0	0	0
Methane	0	0	0	0	2664646		0	571	0
Ammonia	0	23 426	5 124	0	0		0	0	0
Tap water	5272	1	0	2	382	35	44 024	0	1
Additives	171538827	6923	5481	4391	3899407	92910	49465594	679	1390
Biotite	170468	6	1	56	12250	221,215	159032	1	1
Peat production	153977760	5327	5252	2795	2727860	23246,04	7364032	562	664
Sand	17390600	1590	228	1540	1159297	69443	41942530	116	724
Energy	4975795	395	60	1863	55509	8600	7600649	47	76
Transports	17277927	1780	428	7198	1287647	33933	18680644	-332	2206
Biotite	331974	22	5	138	24696	645	358 925	0	42
Coffee waste	23529	4	1	10	1755	48	25 440	-1	3
Products	8590089	1174	287	3 579	640482	17 035	9 287 479	-310	1 097
Horse manure	736387	87	23	307	54764	1 448	796 171	-20	94
Peat	4010524	265	60	1 671	299230	7 794	4 336 121	-1	512
Sand	3583371	234	52	1 493	156569	6 959	3 874 289	1	458
Waste	2052	0	0	1	153	4	2 219	0	0

Appendix 10: Numeric LCIA results for different factors in studied scenarios

Scenario	Sub-scenario	Changed/ additional flow	Result in scenario								
			ADP (fossil) (MJ)	AP (kg SO_2 eq.)	EP (kg PO_4 eq.)	FAETP (kg DCB eq.)	GWP* (kg CO_2 eq.)	HTP (kg DCB eq.)	MAETP (kg DCB eq.)	POCP (kg C_2H_4 eq.)	TETP (kg DCB eq.)
1	a	Ferrous sulphate via synthesis route	70966000	40457	871	25754	1747513	198495	149408974	2145	2364
	b	Ferric sulphate	14972646	8086	153	4687	484971	41008	59745033	435	476
2		Additional polymer	7941318	1295	142	46	317755	1147	6767109	201	6
		Additional ferric sulphate	1749972	942	18	546	56511	4778	6961778	51	55
		Additional electricity	234419	865	182	1 038	38657	8814	11995441	49,6	143
		Recipient, phosphorus (total load)	0	0	29702	0	0	0	0	0	0

Continues on the next page

Continues from the previous page

Scenario	Sub-scenario	Changed/ additional flow	Result in scenario								
			ADP (fossil) (MJ)	AP (kg SO_2 eq.)	EP (kg PO_4 eq.)	FAETP (kg DCB eq.)	GWP* (kg CO_2 eq.)	HTP (kg DCB eq.)	MAETP (kg DCB eq.)	POCP (kg C_2H_4 eq.)	TETP (kg DCB eq.)
3*		Oxygen for ozone	22661971	5986	561	4915	2120864	132472	258536813	380	1906
		PAC	160260824	28032	2657	8778	11526193	547710	910838710	1888	6353
		Additional electricity (does not include scenario 2)	4062844	14998	3151	18 004	669987	152767	207900044	851	2 608
4		Bio-ethanol	23758739	5803	5299	39969	1930372	121743	56088574	113	1246
5	a, b & c	Calcium hydroxide (total result)	3912231	323	64	396	1370618	9776	13411636	14	3651
	a, b &c	Methanol (total result)	86530232	2927	361	67935	2054448	10188	53766844	492	76
	b	WWT, nitrous oxide emissions (total result)	0	580	29244	0	28538380	1391	0	32,5	0
	c	WWT, nitrous oxide emissions (total result)	0	580	23555	0	22954830	1391	0	32,5	0

Appendix 11: Normalized Results of LCA

Impact potential	Viikinmäki process	Metsäpirtti process	Scenario 3
ADP (elements)	5.88×10^{-9}	2.21×10^{-9}	3.82×10^{-8}
ADP (fossil)	3.53×10^{-6}	5.55×10^{-6}	1.52×10^{-5}
AP	2.59×10^{-6}	1.94×10^{-6}	7.80×10^{-6}
EP	1.93×10^{-5}	1.35×10^{-6}	2.02×10^{-5}
FAETP	4.18×10^{-7}	6.64×10^{-8}	6.54×10^{-7}
GWP (excl. biogenic carbon)	7.36×10^{-6}	4.13×10^{-6}	1.47×10^{-5}
HTP	2.10×10^{-7}	2.78×10^{-7}	2.22×10^{-6}
MAETP	1.55×10^{-6}	1.72×10^{-6}	3.52×10^{-5}
ODP	8.33×10^{-8}	9.98×10^{-16}	8.33×10^{-8}
POCP	2.30×10^{-6}	5.43×10^{-7}	4.81×10^{-6}
TETP	6.16×10^{-8}	3.28×10^{-8}	1.94×10^{-7}

Appendix 12: Sensitivity of LCA results in EP, GWP and ADP (fossil) with 5% standard deviation of selected parameters

Parameter	Impact category	1–standard deviation (%)	1+standard deviation (%)
N to the sea from Viikinmäki	ADP (fossil)	0.00	0.00
	EP	-2.50	2.50
	GWP	0.00	0.00
N_2O emissions from Viikinmäki	ADP (fossil)	0.00	0.00
	EP	-0.33	0.33
	GWP	-2.07	2.07
N_2O emissions from Metsäpirtti	ADP (fossil)	0.00	0.00
	EP	-0.18	0.18
	GWP	-1.13	1.13
Methanol production	ADP (fossil)	-1.54	1.54
	EP	-0.01	0.01
	GWP	-0.20	0.20
Peat production	ADP (fossil)	-2.48	2.48
	EP	-0.07	0.07
	GWP	-0.25	0.25
Sand production	ADP (fossil)	-0.33	0.33
	EP	0.00	0.00
	GWP	-0.12	0.12

Appendix 13: Results of the Monte Carlo Analysis

Result category	EP	ADP (fossil)	GWP 100 years (excl biogenic carbon)
Basis scenario	3,82E+05	3,19E+08	6,00E+07
Mean value	3,81E+05	3,20E+08	6,02E+07
Standard deviation	2,46 %	2,57 %	2,17 %
10% percentile	3,71E+05	3,10E+08	5,91E+07
25% percentile	3,76E+05	3,15E+08	5,94E+07
Median	3,82E+05	3,20E+08	6,05E+07
75% percentile	3,87E+05	3,26E+08	6,13E+07
90% percentile	3,93E+05	3,31E+08	6,17E+07
-10%...-8%	2 %	0 %	0 %
-8%...-6%	0 %	2 %	2 %
-6%...-4%	0 %	2 %	2 %
-4%...-2%	18 %	14 %	4 %
-2%...0%	34 %	22 %	34 %
0%...2%	30 %	36 %	30 %
2%...4%	12 %	16 %	28 %
4%...6%	0 %	6 %	0 %
6%...8%	4 %	2 %	0 %
8%...10%	0 %	0 %	0 %

Appendix 14: Carbon Footprint Calculation Results for Viikinmäki Processes

Process		Carbon footprint (ton CO_{2e}/year)	Share of total carbon footprint (%)
Total carbon footprint		43 612	
WWT		38 801	89
	N2O emissions	27 906	64
	CH4 emissions	7 412	17
	Methanol respiration	3 482	8
Sludge treatment at WWTP		99	0
Chemicals		2 482	6
	Calcium hydroxide	497	1
	Methanol	1 884	4
	Polymer	101	0
Screenings and sand		5	0
Energy production and use		739	2
	N2O and CH4 emissions from biogas use	689	2
	Heat production with oil at the plant	47	0
	Solar energy production	3	0
Transports		1 090	2
	External sludge	415	1
	Chemicals	548	1
	Waste	3	0
	Sludge	95	0
Effluent indirect N2O emissions		425	1

Appendix 15: Carbon Footprint Calculation Results for Metsäpirtti Processes

Process		Carbon footprint (ton CO_{2e}/year)	Share of total carbon footprint (%)
Total carbon footprint		10 301	
Composting and soil production		1 745	17
	Loading	44	0
	Composting	1 394	14
	Soil production	946	9
	Use of compost and soil products	-26	0
	Carbon sequestration	-613	-6
Soil additives		6 981	68
	Sand	1	0
	Peat	6 980	68
Energy use		878	9
Transports		697	7
	Sludge from Suomenoja WWTP	29	0
	Soil additives	358	3
	Products	310	3

Appendix 16: Calculated Changes to the Carbon Footprint Results with Optional Parameter Values

Process	Changed parameter	Initial carbon footprint of the process (tons CO_2e/a)	Range of carbon footprint with changed parameter (tons CO_2e/a)
WWT	Use of calculated emission values instead of measurements	38 801	22 983
Energy production and use	CH_4 emission factor from incineration of biogas in gas engine, options in the tool	739	622–7 446
	N_2O emission factor from incineration of biogas in gas engine, options in the tool	739	731–3 690
Effluent indirect emissions	CH_4 emission factor from effluent, options in the tool	425	3 831
Composting and soil production	Carbon sequestration factor, options in the tool	1 745	1 132–2 358
Soil additives	Emission factor of peat production & peat substitution ratio, options in the tool	6 981	5 759–20 888
Total carbon footprint for Viikinmäki and Metsäpirtti	All parameters above	53 914	36 136–77 620